

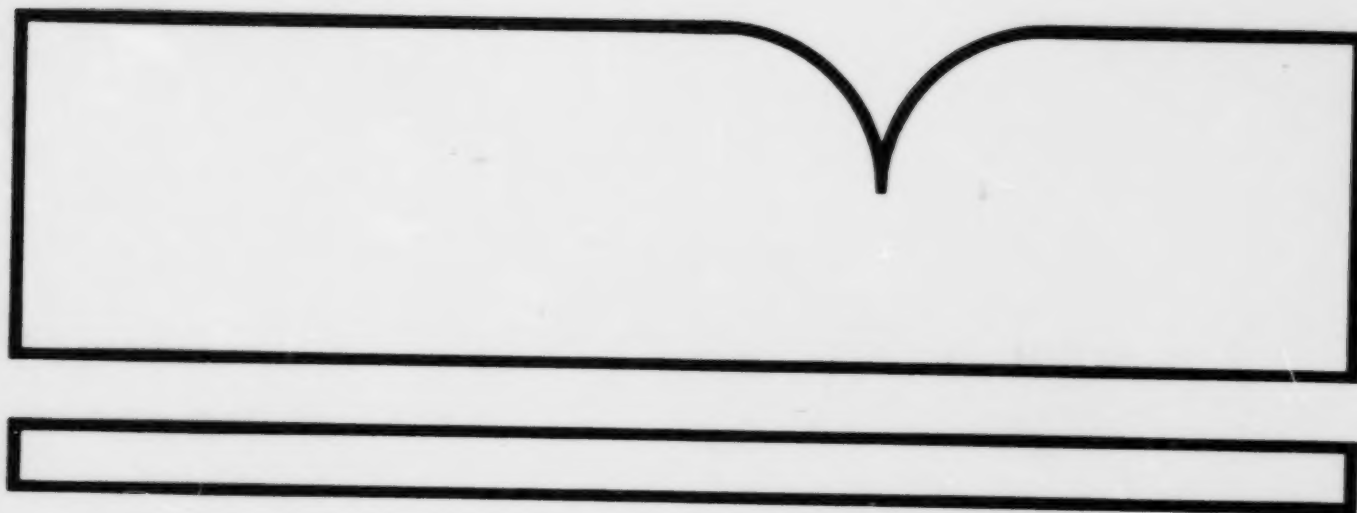
Wastewater Solids Utilization on
Land Demonstration Project

Ocean County Sewerage Authority
Bayville, NJ

Prepared for

Municipal Environmental Research Lab.
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Aug 80



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EPA-600/2-80-090
August 1980

WASTEWATER SOLIDS UTILIZATION
ON LAND DEMONSTRATION PROJECT

by

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Bayville, New Jersey 08721

and

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Grant No. S 801871

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FOREWORD

The Environmental Protection Agency was created because of increasing public and government concern about the dangers of pollution to the health and welfare of the American people. Noxious air, foul water, and spoiled land are tragic testimony to the deterioration of our natural environment. The complexity of that environment and the interplay between its components require a concentrated and integrated attack on the problem.

Research and development is that necessary first step in problem solution and it involves defining the problem, measuring its impact, and searching for solutions. The Municipal Environmental Research Laboratory develops new and improved technology and systems for the prevention, treatment, and management of wastewater and solid and hazardous waste pollutant discharges from municipal and community sources, for the preservation and treatment of public drinking water supplies, and to minimize the adverse economic, social, health, and aesthetic effects of pollution. This publication is one of the products of that research; a most vital communications link between the researcher and the user community.

Using spent solids on land for its nutrient value is not new. The additions of raw waste as a fertilizer has been practiced extensively for centuries in most underdeveloped nations. However, the applications of municipal digested wastewater solids has been practiced on a relatively small scale in this country and only for a few generations. The ever-increasing population and residues from higher degrees of treatment each contribute to a solids disposal problem which is already difficult and complex. Ultimate solids disposal is no longer a matter of simple disposal by "dilution" or "reuse" but one of coping with a series of interacting environmental limitations.

This investigation was undertaken for the purpose of determining the feasibility of utilizing wastewater solids on sandy coastal plain soils in conjunction with environmental considerations such as groundwater quality, odors, heavy metal distribution and wildlife feeding on amended crops needed resolution so that reuse of digested sludge could be placed in proper perspective.

The cost of applying wastewater solids to land will have a considerable bearing on whether or not reuse becomes a viable alternative to other practices for ultimate disposal. Success may be measured on how little it costs (or how little you lose) to meet this first priority. Solving the ultimate disposal problems has enormous value to society. Recovering as much of the costs as possible has a salutary effect. However, to set one's goals only as providing a reuse product equivalent to or competitive with an artificial fertilizer can be misleading and defeating. Initially, the two products are totally different in composition and/or costs per unit weight to produce. Secondly, the waste solids represent a burden which is growing continuously and must be resolved profitably or not.

This report represents a condensation of a much larger effort. Approximately 250,000 separate measurements were made over a four-year period. Those interested in additional detail may wish to examine other data collected on this project. Inquiries concerning the availability of such data should be made directly to the Ocean County Sewerage Authority.

It is hoped that this report will provide the reader with additional insight into (a) the permissible loadings on sandy type soil, (b) techniques for monitoring crop growth, (c) soil changes, (d) groundwater quality, (e) groundwater movement, (f) heavy metal uptake, (g) wildlife feeding, (h) aerosol formation, and (i) crop management considerations. Although recommendations are made for additional study much valuable information is offered to those interested in evaluation and control of wastewater solids application on land.

This report presents results of a study that was conducted to identify alternative methods of treating and disposing of wastewater sludge. It provides information about the beneficial effect of using sewage sludge on sandy soils with low productivity. Insight into permissible application rates, techniques for monitoring crops, effects on groundwater quality, crop responses, wildlife response, and effect on air quality are also provided.

ABSTRACT

This report is a condensed version of a much larger study and presents in summary form the guidelines, limitations, and environmental changes associated with land applications of digested municipal wastewater solids under the conditions of the study. The environmental impact on different soils at various application rates was investigated. The study was conducted over a four-year period, but sludge was applied during the first three years.

Some of the salient considerations were the effect of sludge on groundwater quality, groundwater movement, crop production, wildlife, microbial aerosols, odor, soil properties, crop management, heavy metal uptake and monitoring needs.

This report was submitted in fulfillment of Grant No. S-801871 by The Ocean County Sewerage Authority under the partial sponsorship of the Environmental Protection Agency. The report covers the period June, 1972 to October, 1976.



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Policy and scope of overall effort were defined by the project's Coordinating Committee which consisted of representatives from the major contributors to the project study.

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There were many additional people too numerous to mention who spent time on the project in various capacities.

SECTION 1

INTRODUCTION

BACKGROUND

The disposal of wastewater sludges is probably the most complex problem with which the scientist, engineer, environmentalist and treatment plant operator is faced today in the field of wastewater management. This problem is being rapidly compounded as the volumes of wastewater solids continue to increase. Many communities must upgrade existing treatment facilities or implement new facilities to comply with current Federal and State regulations requiring higher degrees of wastewater treatment. Secondary treatment doubles the amount of wastewater solids generated by primary treatment alone. Adding advanced wastewater treatment produces 25 percent more wastewater solids than secondary treatment alone. Coincident with the increase in volume of wastewater solids is the constraints from more stringent disposal regulations, increased costs, and increasing concern for adverse environmental impacts of conventional disposal methods.

For example, recent claims of beach contaminations and massive fish kills have made it difficult to continue with ocean disposal. The scarcity of large tracts of land and the potential of contamination of the underlying groundwaters have limited the availability of sites for landfill operations. Stringent air pollution control requirements coupled with the high cost of fuel have significantly curtailed the use of incinerators. It has become imperative that we find alternate means of wastewater solids disposal that are environmentally suitable and economically possible.

This project was conceived to demonstrate that wastewater solids could be disposed of via a reuse method in an environmentally acceptable manner. Specifically, the project was conceived to demonstrate that liquid digested domestic wastewater sludge can be utilized as a soil conditioner and fertilizer for the cultivation of crops and enhancement of native vegetation in marginally-productive coastal lands. The usefulness of the enriched native vegetation and crop production as a source of wildlife feed was also evaluated.

A principal objective of the project was to determine optimum loading rates of wastewater solids that can be applied to the land contingent upon the effects of such loading on the underlying groundwater quality. Accordingly, a major section of this report was devoted to defining groundwater contamination and establishing groundwater contamination levels which might be considered for development of groundwater quality standards throughout the Atlantic Coastal area of the United States.

The project was conducted in Ocean County, New Jersey which is a northeastern coastal area of the country. The sandy soils predominating in the coastal area are generally low or practically void of nutrient and organic content and as such are only of marginal value for agricultural purposes. In order to make the project broadly based, and thereby be applicable to other coastal areas of the country, three distinct types of soils were selected for the demonstration. The soils selected are considered to be common to the eastern coastline of the country.

The project commenced in July 1972 with plot preparation followed by well installation. The first application of wastewater solids was made in the spring of 1973. Wastewater solids applications were continuously made over the growing seasons for a three-year period terminating in the fall of 1975. To determine the residual effects of these applications, groundwater monitoring and analyses continued another full year terminating in late fall of 1976.

PROJECT OBJECTIVES

The primary objectives of the project were as follows:

1. To determine the effects on groundwater quality of using soil as a disposal medium for wastewater solids, and to determine groundwater pollutional restraints based upon hydrologic considerations of the sites selected for disposal of solids.
2. To determine the quantity and frequency of wastewater solids application, and the length of time that wastewater solids may be applied to the soil without undesirable effects.
3. To evaluate the prolonged use of soil as a recipient of sludge and determine how it affects both groundwater and soil composition.
4. To determine if disposal of wastewater solids on marginally productive lands could result in an increase in productivity of such lands.

5. To determine the effects of solids application on native vegetation and crop production.
6. To observe wildlife reactions to grazing of crops grown under this process.
7. To determine effects of wastewater solids applications on air quality.
8. To determine the need for monitoring systems and other restraints within the area of the application.

PROJECT PARTICIPANTS

This demonstration project was developed as a joint multidisciplinary project between the following participating agencies:

1. Ocean County Sewerage Authority
2. Rutgers - The State University of New Jersey
3. State of New Jersey, Department of Environmental Protection
4. U.S. Environmental Protection Agency
5. U.S. Department of the Interior-Geological Survey

Technical direction, policy matters, and overall project coordination was delegated to a Coordinating Committee consisting of one representative from each of the participating agencies.

In addition to being involved in the overall development and execution of the project, the participating agencies were responsible for the performance of one or more of the following major functions:

Rutgers University

Site Selection

Analytical determinations

Selection and management of types of vegetation ground cover

Application methods and techniques

Aerosol transport

Groundwater monitoring

The Departments of Environmental Science, Agricultural Engineering and Soils and Crops of the State University participated in this project.

U.S. Geological Survey

Site selection
Subsurface geology
Evaluate changes of soil moisture and water quality
through the unsaturated zone
Chart groundwater flow and evaluate groundwater quality
Rainfall data

State of New Jersey Department of Environmental Protection

Effects on native vegetation
Wildlife reactions

The Divisions of Fish, Game and Shellfisheries and Water
Resources participated in this project.

U.S. Environmental Protection Agency

Grantor of federal grant funds
Project overview

Ocean County Sewerage Authority

Grantee and banker
Project administration

SECTION 2

DISCUSSION OF INITIAL OBJECTIVES AND SUMMARY OBSERVATIONS

The project was undertaken with the expressed purpose of responding to a specific list of objectives. The following summary statements were abstracted from the voluminous data collected since the initiation of the study in July 1972.

Objective 1. To determine the degree of contamination of groundwater from land disposal of wastewater solids, and to ascertain the effects of hydrologic characteristics upon wastewater solids application rates to meet established standards.

The more mobile anions and cations (Na^+ , Ca^{++} , Mg^{++} , Cl^- , SO_4^- , and NO_3^-) show measurable groundwater concentrations above background and control levels directly beneath the application plots. Contamination of the groundwater with heavy metals was generally not detectable, with the possible exception of copper and zinc.

There appears to be no evidence of coliform contamination in any wells due to the application of wastewater solids. A few samples, including some from the control plots, did contain one to four coliforms (per 100 ml) at times.

Upon installation of the wells there was contamination from the solvent used to connect the PVC well casing. After many flushings of the wells, Total Organic Carbon (TOC) concentrations stabilized between 0.0 and 3.0 mg/l. Total Organic Carbon concentrations occasionally exceeded recorded baseline concentrations in most wells. Peak TOC concentrations of 24 mg/l and 20 mg/l were recorded for wells located at the center of application sites.

Groundwater samples for many observation wells showed little or no change in any chemical constituent concentration but in other wells, peak concentrations of many constituents, had increased by more than two (2) orders of magnitude from reported baseline levels. Contaminants were at peak concentrations in winter and spring months when precipitation exceeded evapotranspiration.

The velocity of groundwater flow ranged from 0.62 to 0.69 ft per day at the Webbs Mill sites and 1.1 to 1.6 ft per day at Colliers Mills. At these rates, the affected groundwater will reach the surface system within one year at the Downer soil site and four years at the Woodmansie soil site. The extent of contaminant attenuation between the loading source and the stream must be considered for the purposes of this study. The concentrations of any constituents reaching the stream will be controlled largely by the loadings, attenuation (dilution) and management practices.

For a particular plot, the concentration of the chemical constituent in an observation well was also spatially dependent on the location of the well with respect to the direction of groundwater flow and whether the wells were located within or downgradient of the application site. Those wells designated as "A", "B", and "C" were within the point of application while "D" and "E" wells were 30.5 m (100 ft) and 61 m (200 ft) downgradient of the application plots, respectively. Although Wells "A", "B", and "C" were within an application plot, Well "A" was installed on the upgradient fringe of the plot, Well "B" was in the center, and Well "C" was installed at the far downgradient fringe of the application plot. One would expect little or no change in groundwater quality for Well "A" because uncontaminated groundwater is always flowing through this well. Therefore, the evaluation of groundwater quality from the application of wastewater solid (breakthrough) is best interpreted from the data obtained from Wells "B" and "C". Concentrations of chemical constituents in Wells "B" and "C" within the same plot can also vary significantly due to the heterogeneity of soil; due to clay lenses or coarse sandy strata in the soil profile and the position of the wells relative to the changing direction of flow of contaminants.

The "D" and "E" wells were installed 30.5 m and 61 m downgradient of the application site, according to the first potentiometric maps constructed at the beginning of applications in 1973. However, it was found at a later date that the direction of flow shifted under changing conditions of flow gradients recharge, and discharge. The concentration of chemical composition reported for these wells were affected by the changing flow direction. As a result, the values reported were considerably lower than the representative concentration which lies directly along the line of flow. Additionally, samples (skim samples) from the top of the watertable at downgradient wells are also somewhat diluted because these wells were not subject to vertical breakthrough of sludge-related chemicals, but are still subject to groundwater recharge. The profile sample data were also used to more clearly define the distribution of constituents downgradient from the source plots.

Objective 2. To determine the quantity and frequency of applications, and the length of time that wastewater solids may be applied to the soil without undesirable effects.

Sludge Application on Natural Vegetation Plots

The applications of wastewater solids on natural vegetation plots caused a significant deterioration of groundwater quality beneath the plot. The naturally acid soil conditions (circa pH 4.5) contributed to concentrations of many sludge-related constituents approaching and even exceeding the values reported for the heaviest loaded (89.6 t/ha/y) applications. Contaminant concentration beneath the three natural vegetation plots seem to vary according to soil pH. Groundwater contamination was the greatest beneath the Woodmansie plot where the average soil pH was as low as 4.3. These acid conditions significantly increased the mobility of many sludge constituents, particularly ammonia nitrogen, organic acids (TOC) and even heavy metals. Compounding the effect of the acid soil conditions is the lack of harvesting of a cover crop which would remove many of the sludge constituents from the application site. Therefore, the natural vegetation site becomes a repository for groundwater contaminants.

From these observations it appears that the application of wastewater solids is not suitable for natural vegetation areas that are typical of the New Jersey Pine Barrens if the 44.8 t/ha/y loading rate is utilized without prior pretreatment of soils. Unfortunately, the loading rates were not varied on the natural vegetation sites and therefore it is not possible to establish a loading rate which will result in a minimal deterioration of groundwater.

Sludge Application Rates for Cultivated Crops

It has been reported by Sommers (1) that the determination of annual application rates must be contingent upon the nitrogen and cadmium content of the sludge and the specific crop selected

(1)

Adapted from Sommer, L. E., "Principles of Land Application of Sewage Sludge". In Design Seminar for Sludge Treatment and Disposal, Environmental Protection Agency, Cincinnati, Ohio, 1977.

for cultivation. The number of years of sludge application is limited by heavy metal concentrations. Employing Sommers' formula, calculations were made utilizing OCSA collected data with the following results:

1. Nitrogen Requirements

$$\text{Tons sludge/acre} = \frac{\text{crop requirement} - \text{residual nitrogen}}{\text{pounds available/ton sludge}}$$

Crop nitrogen requirement for double cropping of rye and Midland bermudagrass is approximately 350 lbs/acre. Based on sludge characteristics the pound of available nitrogen per ton sludge ranged from 50 to 63 lbs/ton sludge. Residual nitrogen was calculated as 13 lbs after the 1st year application and 26 lbs after the second year.

Since sludge was surface applied, doubling the allowable rate is indicated because of loss of ammonia nitrogen by volatilization. Thus the nitrogen requirement limits the annual loading rate to a maximum of 10-12 tons/acre (22.4 - 26.9 t/ha/y).

2. Cadmium Annual Limitations

$$\text{Tons sludge/acre} = \frac{2 \text{ lbs Cd/acre}}{\text{ppm Cd} \times .002}$$

Cadmium in the sludge applied averaged 17.9 ppm on a dry basis. Therefore, the annual allowable sludge application rate based on Cd additions would be 56 dry ton/acre.

3. Heavy Metal Limitations

The total amounts of lead, zinc, copper, nickel, and cadmium suggested on agricultural soil from land application of sewage sludge was developed by cooperative efforts of regional research projects on Nc-118, W-124, SEA, and USDA. Table 1 lists the total amounts of metal that can be applied to the soil.

TABLE 1.* TOTAL AMOUNTS OF METAL THAT CAN BE APPLIED TO THE SOIL

Metal	Soil CEC, meq/100g		
	5	5-15	15
	lbs/acre		
Pb	500	1000	2000
Zn	250	500	1000
Cu	125	250	500
Ni	50	100	200
Cd	5	10	20

The suggested limits are based on soil cation exchange capacity (CEC). The soils of Ocean County, New Jersey have a low CEC ranging from 1-5 meq/100g.

The total tons of sludge applied per acre can be calculated from the following equation:

$$\text{Tons sludge/acre} = \frac{\text{total lb metal/acre}}{\text{ppm metal} \times 0.002}$$

The average metal characteristics of Ocean County sludge suggested maximum metal loadings and calculated tonnage of sludge necessary to exceed the allowable loadings is summarized in Table 2.

*

Table developed by cooperative efforts of various regional projects NC-118 and W-124 and SEA, USDA. (1)

TABLE 2. METAL CHARACTERISTICS OF OCEAN COUNTY SLUDGE AND
MAXIMUM ALLOWABLE METAL AND SLUDGE LOADING

	Concentration in Sludge (mg/g)	Maximum Allowable Kilograms of Metal per Hectare	Total Allowable Metric Tons of Sludge per Hectare
Pb	360	560	1557
Zn	2653	280	105
Cu	913	140	155
Ni	43	56	1301
Cd	17.9	5.6	314

From Table 2, it can be seen that sludge application is limited by Zinc (lowest amount) or 47 tons per acre.

In summation, the annual sludge rate based on nitrogen required for cropping rye and Midland bermudagrass was found to be 10-12 tons/acre (22.4 t/ha/y), if surface application techniques are utilized. Additionally, lime and fertilizer potassium (K) would have to be applied to obtain optimum yields. If the application site was continuously cropped and 10 tons/acre/year (22.4 t/ha/y) of sludge were applied, the site could be utilized for approximately 5 years. The 5-year life span would be limited by the total allowable zinc loading. Stripping surface accumulated sludge would appreciably extend the life of the site for sludge application purposes.

Based on groundwater quality data obtained from experimental application plots in Ocean County, New Jersey, the 10 tons/acre/year (22.4 t/ha/y) sludge loading rate does not result in significant contamination of the groundwater quality. Therefore, surface application of wastewater solids at 10 tons/acre/year (22.4 t/ha/y) appears to be environmentally acceptable from a groundwater quality aspect for a period of 5 years.

Other studies indicate that the population of denitrifying soil micro-organisms increases with years of manure added to the soil, so the rate that land could accommodate might increase with time. Nitrifying inhibitors might be applied to reduce groundwater contamination by nitrates.

The surface accumulation after 3 years of application was appreciable (particularly at the highest loading rate) and could be removed for sale as a soil amendment, or incorporated to provide additional nutrients to crops and increase the

nutrient and moisture retention capacity of the soil. If accumulated surface sludge is skimmed off for sale there should be no need to "rest" the site. If a 3-year accumulation is tilled in, sufficient nutrients may be released to warrant cessation of applications until the flush of nitrogen is dissipated.

Initially the 22.4 t/ha (10 TA) rate did not appear to supply sufficient nutrients for good crop production and N deficiencies were evident on the Midland bermudagrass. The 44.8 t/ha resulted in better production of grass, but the 89.6 t/ha did not appear excessive even when applied to cool season perennial grasses so as to inundate regrowth in July. Although this is an example of gross mismanagement it served to illustrate crop tolerance. Crop injury developed when heavy applications were made to grass that was tall enough to harvest as hay crop.

With proper site selection, diversification of cropping systems and logical coordination of sludge applications with overall management, the application of sludge from residential communities at rates of 44.8 t/ha/y or more should be practical. However, further studies are needed.

Objective 3. To Determine How the Prolonged Use of Soil as a Wastewater Solids Receptor Affects Groundwater and Soil Quality

The cumulative impact of wastewater solids application on groundwater quality was studied over a four (4) year period. Application of wastewater solids in Ocean County was initiated in May 1973 and was applied for three (3) years until October 1975. No solids were applied during the late fall and winter months. Groundwater monitoring was also performed during this period and was continued for an additional year through November 1976, thus a complete year of groundwater quality data was obtained after wastewater solids applications were discontinued.

Table 3 summarizes the Relative Yearly Changes in contaminant concentrations for the 12 wastewater solids application plots. Contaminant concentrations beneath the 22.4 t/ha/y application plots initially increased, but then stabilized after two years of wastewater solids applications. At the other extreme, contaminant levels beneath the sandy soil Lakewood 44.8 t/ha/y and 89.6 t/ha/y plots steadily increased over the four-year study. Contaminant concentrations also increased beneath the Woodmansie 44.8 t/ha/y plot during all four years of the project, however the increase was very small and gradual as compared to the breakthrough concentrations observed at the Lakewood plot receiving the same wastewater solids loading.

TABLE 3. RELATIVE YEARLY CHANGES IN CONTAMINANT CONCENTRATIONS BENEATH INDIVIDUAL APPLICATION PLOTS (WELLS 'B' and 'C') FROM 1973 TO 1976

Location	Loading (t/ha/y)	Year			
		1973	1974	1975	1976
Colliers Mill Downer	22.4	slight increase	slight increase	stable	stable
	44.8	slight increase	increase	slight increase	decrease
	44.8#	slight increase	increase	increase	stable
	89.6	slight increase	increase	increase	decrease
Webbs Mill Lakewood	22.4	slight increase	slight increase	increase	stable
	44.8	slight increase	increase	large increase	increase
	44.8#	slight increase	slight increase	slight increase	slight decrease
	89.6	slight increase	large increase	increase	increase
Webbs Mill Woodmansie	22.4 (B)	slight increase	slight increase	stable	stable
	(C)	slight increase	increase	stable	stable
	44.8	slight increase	slight increase	slight increase	slight increase
	44.8#	slight increase	increase	large increase	large increase
	89.6	slight increase	increase	increase	decrease

Natural Vegetation
(B), (C) Well Designation for Well 'B' and 'C' on 22.4 t/ha/y Woodmansie plot.
Stable: No significant change in groundwater quality from previous year.

Beneath all other plots, contaminant breakthrough concentrations increased during the first three years of the study, but decreased or remained stable during the fourth and final year of sampling. Although a decrease in contaminant levels was observed at many plots, in no case did the contaminant concentrations of such sludge-related constituents as chloride, sulfate, calcium and nitrate decrease to background levels.

The application of wastewater solids in Lakewood soils appeared to produce the greatest leaching of contaminants into the groundwater. The Woodmansie soil has the lowest contaminant concentrations probably because of considerable silt and clay development in the lower subsurface and substratum of the soil. However, this is contrary to the expected, because the Woodmansie soil is more acid than Lakewood or Downer soils. Where soil characteristics were similar to that of a coarse sand with minimal clay development, even the low wastewater solids loading rate of 22.4 t/ha/y can cause contaminant leaching. Contaminant leaching according to soil type and solids loading rates were as follows:

Loading Rate t/ha/y	Contaminant Leaching According to Soil Series		
22.4	Lakewood	Downer	Woodmansie
44.8	Lakewood	Downer	Woodmansie
89.6	Lakewood	Downer	Woodmansie

It should be noted however, that the above observations are not similar to findings reported for Well "B" on the 22.4 Woodmansie application plot because this well was inadvertently installed in a sand that is not within the range of characteristics of the Woodmansie soil series. Understandably, this particular well demonstrated contaminant leaching rates almost as high as the levels observed beneath the Woodmansie 89.6 t/ha/y plot.

Contaminant breakthrough concentrations were compared to National-Interim-Primary-Drinking Water Regulations proposed by the U.S. Environmental Protection Agency and the Potable Water Standard established by the State Department of Environmental Protection of the State of New Jersey. These standards were compared to the average concentrations observed at each application plot. A total of 23 groundwater constituents were tabulated according to plot loading rates and soil type. Table 4 presents the average of contaminant breakthrough concentrations for these 23 constituents. Two additional distinctions pertaining to groundwater quality are also identified in Table 4 (1) when the contaminant concentration is at or above the drinking water standard and (2) when the contaminant concentration is within 50% of the maximum limit.

TABLE 4. AVERAGE INCREASE OF CONTAMINANTS IN GROUND WATER SAMPLES
OVER BACKGROUND LEVELS OF SELECTED PARAMETERS

Plot		Parameters					
Location	Loading (t/ha/y)	T.O.C.	NO ₃ -N	NO ₂ -N (Increase in concentration mg/l)	NH ₃ -N (Increase in concentration mg/l)	SO ₄ (mg/l)	CI
Colliers Mills Downer	22.4	0.6	2.0	0.0	0.0	6	5
	44.8	0.4	5.4*	0.0	0.0	28	8
	44.8#	1.1	14.7**	0.0002	0.05	23	15
	89.6	1.9	23.8**	0.0004	0.06	72	25
Webbs Mill Lakewood	22.4	0.1	3.7	0.0	0.02	15	3
	44.8	2.2	22.7**	0.0	0.08	75	17
	44.8#	.4	4.3	0.0	0.0	7	5
	89.6	5.8	54.0**	0.003	0.13	123*	44
Webbs Mill Woodmansie	22.4	2.5	11.6**	0.0002	0.04	52	11
	(B)	0.6	4.1	0.0002	0.0	11	1
	44.8	0.6	5.5*	0.0	0.06	9	3
	44.8#	6.7	26.7**	0.0016	2.20*	63	38
	89.6	.9	13.2**	0.0003	0.03	31	8

Natural Vegetation Plot.
(B), (C) Well Designation for Wells 'B' and 'C' on 22.4 t/ha/y Woodmansie Plot.
** Increase is at or above applicable Drinking Water Standards.
* Increase is within 50% of applicable Drinking Water Standards.

(continued)

TABLE 4. (continued)

Plot		Parameters					
Location	Loading (t/ha/y)	T.D.S.	Hardness (Increase in concentration mg/l)	Ca	Mg	Na	K
Colliers Mills Downer	22.4	19	10	1.4	1.5	2.0	0.0
	44.8	54	37	10.0	1.9	4.3	0.1
	44.8#	87	48	16.0	1.9	7.1	0.5
	89.6	330*	132*	37.0	9.0	10.5	0.3
Webbs Mill Lakewood	22.4	45	18	2.2	2.5	0.9	0.04
	44.8	145	125*	33.0	11.8	7.4	0.34
	44.8#	14	18	5.9	.7	2.0	0.11
	89.6	653*	289**	87.5	17.5	19.4	0.84
Webbs Mill Woodmansie	22.4 (B) (C)	72	70	12.0	9.6	5.7	0.71
	44.8	-	16	1.7	2.8	0.8	0.32
	44.8#	30	22	3.8	3.0	1.7	0.31
	89.6	250* 92	122* 60	41.0 18.0	4.5 3.5	12.0 4.4	2.10 0.37

Natural Vegetation.
 (B), (C) Well Designation for Well 'B' and 'C' on 22.4 t/ha/y Woodmansie plot.
 ** Increase is at or above applicable Drinking Water Standards.
 * Increase is within 50% of applicable Drinking Water Standards.

(continued)

TABLE 4. (continued)

Plot		Parameters					
Location	Loading t/ha/y	B	Fe	Mn (Increase in concentration mg/l)	Al in concentration mg/l)	Cr	Pb
Colliers Mills Downer	22.4	0.0	0.0	0.12	0.0	0.0	0.0
	44.8	0.0	0.0	0.25	1.3	0.0	0.0
	44.8#	0.0	0.0	0.44	3.4	0.0	0.0
	89.6	0.01	0.0	0.66	4.6	0.0	0.0
Webbs Mill Lakewood	22.4	0.0	0.0	0.03	0.6	0.0	0.0
	44.8	0.0	0.01	0.05	4.0	0.0	0.0
	44.8#	0.0	0.0	0.03	1.4	0.0	0.0
	89.6	0.02	0.02	0.13	8.0	0.0	0.0
Webbs Mill Woodmansie	22.4 (B)	0.0	0.0	0.08	0.6	0.0	0.0
	(C)	0.0	0.0	0.05	2.0	0.0	0.0
	44.8	0.0	0.0	0.05	0.1	0.0	0.0
	44.8#	0.01	0.08	0.33	11.5	0.0	0.0
	89.6	0.0	0.01	0.10	1.6	0.0	0.0

Natural Vegetation.

(B), (C) Well Designation for Well 'B' and 'C' on 22.4 t/ha/y Woodmansie plot.

** Increase is at or above applicable Drinking Water Standards.

* Increase is within 50% of applicable Drinking Water Standards.

TABLE 4. (continued)

Plot		Increase in concentration mg/l)				
Location	Loading (t/ha/y)	Cu	Zn	Ni	Cd	Hg
Colliers Mills Downer	22.4	0.0	0.02	0.0	0.0006	0.0
	44.8	0.002	0.03	0.0007	0.0004	0.0
	44.8#	0.001	0.07	0.0017	0.0013	0.0
	89.6	0.002	0.05	0.0028	0.0012	0.0
Webbs Mill Lakewood	22.4	0.001	0.01	0.0002	0.0003	0.0
	44.8	0.001	0.06	0.0005	0.0005	0.0
	44.8#	0.0	0.03	0.0	0.004	0.0
	89.6	0.160	0.17	0.0036	0.0007	0.0
Webbs Mill Woodmansie	22.4 (B)	0.0	0.04	0.0008	0.0005	0.0
	(C)	0.0	0.02	0.0003	0.0004	0.0
	44.8	0.0	0.01	0.0007	0.0002	0.0
	44.8#	0.004	0.25	0.0032	0.0031	0.0
	89.6	0.0	0.02	0.0018	0.0007	0.0

Natural Vegetation.
 (B), (C) Well Designation for Well 'B' and 'C' on 22.4 t/ha/y Woodmansie plot.
 ** Increase is at or above applicable Drinking Water Standards.
 * Increase is within 50% of applicable Drinking Water Standards.

Heavy metals did not appear to have a significant impact on groundwater quality. The average breakthrough concentrations of all heavy metals were generally equivalent to or slightly above the natural background levels. When a marginal increase in metal concentration was observed, particularly for zinc and copper (refer to the Lakewood 89.6 t/ha/y and Woodmansie natural vegetation plots), it was well below the maximum contaminant limits for potable water.

The averages of contaminant concentrations beneath individual plots were compared with potable drinking water standards. The peak concentrations were not considered in this comparison. It appeared that the average contaminant concentration was more representative of the long-term response of the groundwater system to contaminant leaching, while the peak concentration was only the maximum level observed at only one single point in time.

Specific organic compounds were not identified in groundwater samples. Only a coarse measurement of total organic carbon was obtained for each sample. Total organic carbon was found to have increased above the background level beneath many of the plots (Table 4). The breakthrough of organic carbon as measured by TOC tends to indicate that trace organics may have entered the groundwater. This is particularly true for those plots receiving the higher wastewater solids loadings, Lakewood 89.6 t/ha/y plot and where soil conditions were extremely acid (as was the case with all the natural vegetation plots).

Nitrate-nitrogen was the limiting groundwater chemical constituent with respect to drinking water standards. Additionally, average concentrations of total dissolved solids, chlorides, sulfates, and total hardness (calcium and magnesium) were approaching, and in many cases exceeding the recommended limit beneath the following locations: (a) plot receiving 89.6 t/ha/y of wastewater solids, (b) natural vegetation plots (Downer and Woodmansie), and (c) where soil characteristics were similar to that of a coarse sand. For the Downer and Woodmansie soils, it would appear a surface application of wastewater solids loading rate in excess of 22.4 t/ha/y and possibly as much as 44.8 t/ha/y could be applied without violating the potable water standards downgradient from the application site. In the case of the more sandy Lakewood soil, the maximum allowable loading rate appears to be 22.4 t/ha/y. However, where the soil is characteristic of a coarse sand, containing little or no silt or clay lenses, even the 22.4 t/ha/y application rate could violate potable drinking water standards. However, the application rates suggested are based on contaminant levels directly beneath the application plots. They do not reflect the possible dilution downgradient

of the application plot from the natural diffusion and dispersion of the contaminant in the groundwater. A buffer area between the application site and the nearest potable water source would provide a chance for dilution to take place.

Acidification of the surface increments of mineral soil was related to rate of sludge applied. This was not anticipated since the Ca content of the sludge was approximately 2.5%. Apparently, liming, as well as K fertilization, is needed on sludge-treated land and should be related to the rate of sludge applied.

Some movement of elements and changes of soil pH through the profile of these sandy soils was apparent. These changes appear to be within tolerable limits over the time of this study. No increase in organic matter in the upper soil strata could be measured with the techniques employed, however, increased dark color in these soils was observed to correlate with the loading rates. Lack of measurable increase in soil cation exchange capacity may relate to the still relatively raw, coarse nature of the organic matter of the sludge that entered the soil. Most of the sludge remained on the soil surface.

Objective 4. To Determine the Effects of Wastewater Solids Application on Land Productivity

Plant nutrients supplied by the stabilized domestic sludge applied to the soil surface proved to be appreciable. No detrimental effects of heavy metals on plant growth were detected.

Both quantity and quality of plants produced were enhanced by sludge applications. The elements, N, P, Ca, Mg, Cu, and Zn usually increased in grasses with rate of sludge applied.

Yields of grasses were difficult to determine accurately because of losses attributable to wildlife grazing. Loss of plant production also occurred when sludge was applied when grass was so tall that lodging resulted and only the upper leaves were harvested. Improperly timed applications also reduced the fertilizer efficiency of applied nutrients. Low K supply in sludge undoubtedly limited plant production.

Although crop production with sludge application was greatly increased, it is felt that the full soil amelioration potential was not realized because the bulk of the sludge remained on the surface. The surface application technique was considered best for this short term study from the point of view of maximizing volatilization loss of N compounds, minimizing groundwater contamination, saving the cost of incorporation, and avoiding crop disturbance and soil erosion loss.

The surface accumulation after 3 years of application was appreciable (particularly at the highest loading rate) and could be removed for sale as a soil amendment, or incorporated to provide additional nutrients to crops and increase the nutrient and moisture retention capacity of the soil. If accumulated surface sludge is skimmed off for sale, there should be no need to "rest" the site. If a 3-year accumulation is tilled in, sufficient nutrients may be released to warrant cessation of applications until the flush of nitrogen is dissipated.

It became apparent, however, that there are many opportunities to improve on the system of land application that was employed in this study. To increase crop production and removal of potential pollutants, coordination of applications and crops management must be maximized. Diversification of crops, particularly with regard to cool and warm season species would facilitate application and would better utilize nutrient elements in sludge and decrease groundwater contamination. The major NO_3 breakthrough occurred in this study where the heaviest rate was applied to the sandiest soil in the final year that had the highest rainfall on record. This occurred in the season, January through March, when there was little or no crop growth. Also, as much as 40% of the year's application had been made after the last harvest of the major crop, Midland bermudagrass. If a non-dormant type cereal rye had been available, the breakthrough should have been much reduced. Greater wildlife utilization could have been realized if more cool season crops had been available. Other studies indicate that the population of denitrifying soil microorganisms increases with years of manure added to the soil, so the wastewater solids application rate that land could accommodate might increase with time. Nitrifying inhibitors might be applied to reduce groundwater contamination by nitrates.

Objective 5. To Determine the Effects of Wastewater Solids Applications on Native Vegetation and Crop Production

Both quantity and quality of plants produced were enhanced by sludge applications. The elements N, P, Ca, Mg, Cu, and Zn usually increased in grasses with rate of sludge applied. Nitrogen and P contents of leaf tissue of several woody species were increased significantly by sludge applications. Some increase in K, Mg, Zn, Cu, and Pb was noted in these tissues. Calcium and Mn actually decreased in foliage of woody species treated with sludge. Wood species in native stands increased in trunk or twig growth in response to sludge applications as measured one and one half years after initiation of sludge applications.

Yields of grasses were difficult to determine accurately because of losses attributable to wildlife grazing. Loss of

The surface accumulation after 3 years of application was appreciable (particularly at the highest loading rate) and could be removed for sale as a soil amendment, or incorporated to provide additional nutrients to crops and increase the nutrient and moisture retention capacity of the soil. If accumulated surface sludge is skimmed off for sale, there should be no need to "rest" the site. If a 3-year accumulation is tilled in, sufficient nutrients may be released to warrant cessation of applications until the flush of nitrogen is dissipated.

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Yields of grasses were difficult to determine accurately because of losses attributable to wildlife grazing. Loss of

plant production also occurred when sludge was applied when grass was so tall that lodging resulted and only the upper leaves were harvested. Improperly timed applications also reduced the fertilizer efficiency of applied nutrients. Low K supply undoubtedly limited plant production.

Our data indicated that the sludge-stimulated crop of Midland bermudagrass depleted soil moisture more rapidly than the no-sludge crop. Crop production could have been increased further and the land's capacity for waste disposal could have been augmented if it were irrigated with effluent. This would be a logical procedure if pipeline transport were available. Alternately, irrigation from on-site wells would have been beneficial from the point of view of increasing crop production and increasing N-loss through denitrification. Liming and K fertilization would then be more important.

Initially the 22.4 t/ha/y (10 TA) rate did not appear to supply sufficient nutrients for good crop production and N deficiencies were evident on the Midland bermudagrass. The 44.8 t/ha rate resulted in better production of grass, but the 89.6 t/ha did not appear to inhibit grass growth even when applied to cool season perennial grasses so as to inundate the grass in July. This is gross mismanagement but served to illustrate crop tolerance. Crop injury developed when heavy applications were made to grass that was tall enough to harvest as a hay crop.

Objective 6. To Observe the Effects of Wastewater Solids Applications on Wildlife Reactions

Elements reportedly toxic to animals didn't increase in the plant tissues to presumed injurious levels. Animal acceptance of sludge-treated grass was good, and a clearcut preference by deer and rabbits was seen for grass treated with the highest rate of sludge. Wildlife consumption of Kentucky bluegrass and cereal rye in fall and spring and sudangrass in summer was appreciable. Deer and rabbits consumed little Midland bermudagrass, but cattle readily ate it even when obviously coated with sludge.

In less than 24 hours after application, deer tracks were observed over tire tracks of the applicator equipment, hence, it was concluded that deer were not repelled by application of fresh sludge.

Objective 7. To Determine Any Air Pollution Effects in Regard to Odor and Bacterial Aerosol Transmission During Solids Applications

Odors at the sludge disposal sites were moderate to slight under most conditions as long as well digested sludge was used.

The worst odors were appreciably less offensive than those commonly associated with farm manures.

The maximum distances (panel medians) downwind from the disposal sites at which members of two panels could still detect any odor were 35 and 39 m (114 and 127 ft) immediately prior to spraying on two different days. The comparable values during and immediately after spraying were 156 and 202 m (512 and 661 ft). One especially sensitive panel member detected odors from the site at a maximum distance of 219 m (718 ft).

The predominant odors at the sites appeared to be derived directly from the sludge, rather than from its decomposition at the site. This was reflected in the odor descriptions, which included stale, earthy, musty, and moldy.

The major emphasis in odor control must be placed on the prevention of odor formation. This includes the necessity for using only well-digested sludge, as well as limiting the application rate to prevent flooding and anaerobic conditions.

The potential for odor complaints may be further reduced by providing a sufficient buffer zone between the site and any unwilling receptors. A minimum distance of $\frac{1}{2}$ km ($\frac{1}{4}$ mile) would appear desirable for the surface application method.

Aerosol Transmission

Sampling under the given field conditions and using the total plate count on TGE Agar, it was difficult to demonstrate statistical differences between samples.

Microbial aerosol concentrations in the absence of fresh spraying (before a daily application) were virtually identical to the background counts observed prior to the first sludge application, averaging 200 per cubic meter (5-7 per cubic foot). This value is in keeping with counts reported in the literature. Two exceptional days with very high counts presumably resulted from the sporulation of saprophytes growing on the sludge plots.

At the edge of the plots, microbial aerosol concentrations increased during spraying to an average value of 3,000 per cubic meter (86 per cubic foot). After spraying the counts decreased rapidly, but were still elevated compared to the background readings (average, 490 per cubic meter or 14 per cubic foot).

Averaging concentrations per cubic meter (per cubic foot) during spraying increased to 410 (11.6) at 15 m (50 ft) and 190 (5.4) at 30 m (100 ft) downwind, but the 30 m reading could

not be shown to be significantly different than the before spraying "background" of 130 (3.7). Samples collected at these distances 5-30 minutes after spraying had returned to background levels.

Concentrations in samples collected on top of the sludge spraying apparatus, to which the operator was exposed, were similar to those collected during spraying at the edge of the site, averaging 2,600 per cubic meter (73 per cubic foot).

Overall, the percentage of large particles (5mm) increased slightly during spraying (70%) relative to before and after spraying (63% each). Only about 9% of the particles were 1-2 mm in diameter, the size range of greatest deposition in the alveoli.

There is a need for a suitable indicator for microbial aerosols produced by spray application of sewage sludge on land.

The potential for adverse effects from microbial aerosols produced by the spray application of sewage sludge on land is difficult to evaluate. Generally, concentrations appear to diminish rapidly with distance, becoming only marginally higher than background readings at 30 m (100 ft) under most conditions. Sporulation of saprophytes on the sludge plots is still readily discernible at 30 m (100 ft) however, (1,700 counts per cubic meter or 47 per cubic foot before spraying, down from 3,900 or 100 at 15 m (50 ft)). Thus, an adequate buffer zone must be considerably larger than 30 m (100 ft). Dispersal by migratory animals, such as birds and insects, must also be considered.

Objective 8. To Recommend Monitoring Systems and Other Restraints Within the Area of the Application Sites

1. Fences, barriers, or remoteness of site can protect the public from exposure to sites where sludge is applied to the soil surface.
2. Groundwater movement should be determined prior to plot layout.
3. Soil pH must be maintained above 6.5 by liming to minimize movement of heavy metals.
4. The groundwater hydrology, area of application, loading rate, and type of soil will determine the number of monitoring wells which should be placed above, within and downgradient of the application site to identify and characterize potential groundwater contamination.

5. Downgradient wells should be within 30.5 m (100 ft) of the application site.
6. The ideal monitoring scheme should include both static and dynamic sampling; i.e., skim-type and profile samples for static analysis of the groundwater system and pumped samples to determine the dilution effects.
7. Thorough groundwater sampling should be conducted at least four (4) times a year.
8. Groundwater analysis should include the following parameters whenever possible.

Total Coliforms
 Total Dissolved Solids
 Ammonia-nitrogen ($\text{NH}_4\text{-N}$)
 Nitrate-nitrogen ($\text{NO}_3\text{-N}$)
 Nitrate-nitrogen ($\text{NO}_2\text{-N}$)
 Total Phosphate ($\text{PO}_4\text{-P}$)
 Orthophosphate ($\text{O-PO}_4\text{-P}$)
 Alkalinity
 Hardness
 Total Organic Carbon (TOC)
 Chloride (Cl)
 Fluoride (F)
 Metals (Hg, Zn, Mn, Fe, Mg,
 Pb, Cd, Cu, Ni, Al)

Temperature
 pH
 Turbidity
 LAS or Alkyl
 Sulfates
 Specific Conduct-
 ance
 Silica (SiO_2)
 Potassium (K)
 Sodium (Na)
 Calcium (Ca)
 Sulfate (SO_4)
 Boron (B)

SECTION 3

RECOMMENDATIONS

1. Based upon conservative considerations (excluding dilution and diffusion downgradient) data from wells located adjacent to application areas warrant the following:
 - a. A loading of 22.4 t/ha/y (10 T/A/yr) of municipal digested wastewater solids is feasible for sandy (Lakewood) nutrient deficient soils provided pH control is practiced.
 - b. This loading may be increased to as much as 44.8 t/ha/y (20 T/A/yr) on soils containing sandy loam (Woodmansie and Downer soils).
2. To maximize crop yields sludge application should be supplemented by potash (K) additions.
3. Good crop management procedures can and will increase the quality and quantity of plants produced. Further investigations of soil management techniques such as surface removal and/or plowing of surface cake should be made.
4. Sludge applications immediately before harvesting grass should be avoided to prevent inundating and contaminating the crop.
5. Irrigation may be considered for maximizing crop production on sandy soils.
6. Rates of annual application of municipal digested solids appear to be limited more by concern for groundwater quality than for crop production. Heavy metal uptake by plants was not a problem.
7. Because wildlife will consume sludge amended crops readily, measures such as fencing will be needed to protect such growth areas should harvesting be contemplated.
8. If odors and aerosols are unacceptable outside of the disposal site, a $\frac{1}{4}$ mile wide buffer should be provided when surface application of liquid digested sludge is planned.

9. Monitoring should consider at a minimum the following:

- a. Number and location of monitoring wells must be based on study of the hydrology, soil type, and sludge loading rates.
- b. Both static and dynamic sampling procedures.
- c. Periodic groundwater sampling and complete sludge, crop, soil, and water analyses.

SECTION 4

SITE SELECTION FOR APPLICATION OF WASTEWATER SOLIDS

LOCATION AND DESCRIPTION

The wastewater solids utilization project comprises three sites in the Pine Barren Region of Ocean County, New Jersey (Figure 1). The Downer Soil site, at Colliers Mills, lies about 1,158 m (3,800 ft) north of Colliers Mills in Jackson Township (Figure 2). The Lakewood and Woodmansie Soil sites, Webbs Mill sites 1 and 2, located about 24.13 km (15 mi) south of Colliers Mills, lie within 762 m (2,500 ft) of each other in Lacey and Manchester Townships (Figure 3). The sites are isolated from populated areas and lie within the boundaries of land managed by the New Jersey State Division of Fish, Game and Shell Fisheries.

The topography of the sites is one of low relief typical of the Coastal Plain. The Downer soil site, at Colliers Mills is at an altitude between 42.7 and 45.7 m (140 and 150 ft), and slopes towards a marshy stream at an altitude of about 39.6 m (130 ft). The Lakewood and Woodmansie soil sites at Webbs Mill are at an altitude between 38.1 and 44.2 m (125 and 145 ft) located in the interfluvium between Webbs Mill Branch of Cedar Creek and an unnamed tributary of Webbs Mill Branch. The confluence of these shallow streams lies at an altitude of about 36.6 m (120 ft) and east of the sites. All three of the sites are immediately east of the major drainage divide, separating the Delaware River drainage from the Atlantic Ocean drainage (Figures 2 and 3).

The climate of the area is temperate. Long-term records indicate an annual average temperature of 53°F (12°C) with precipitation of about 114.3 cm (45 in) at Colliers Mills and 116.8 cm (46 in) at the Webbs Mill sites.

Evaporation and transpiration, or the losses of water to the atmosphere from wetted surfaces or through plant processes, are about 61.0 to 63.5 cm (24 to 25 in).

GEOLOGY

The sites are underlain by three different soil types which occur extensively in the outer Coastal Plain region from

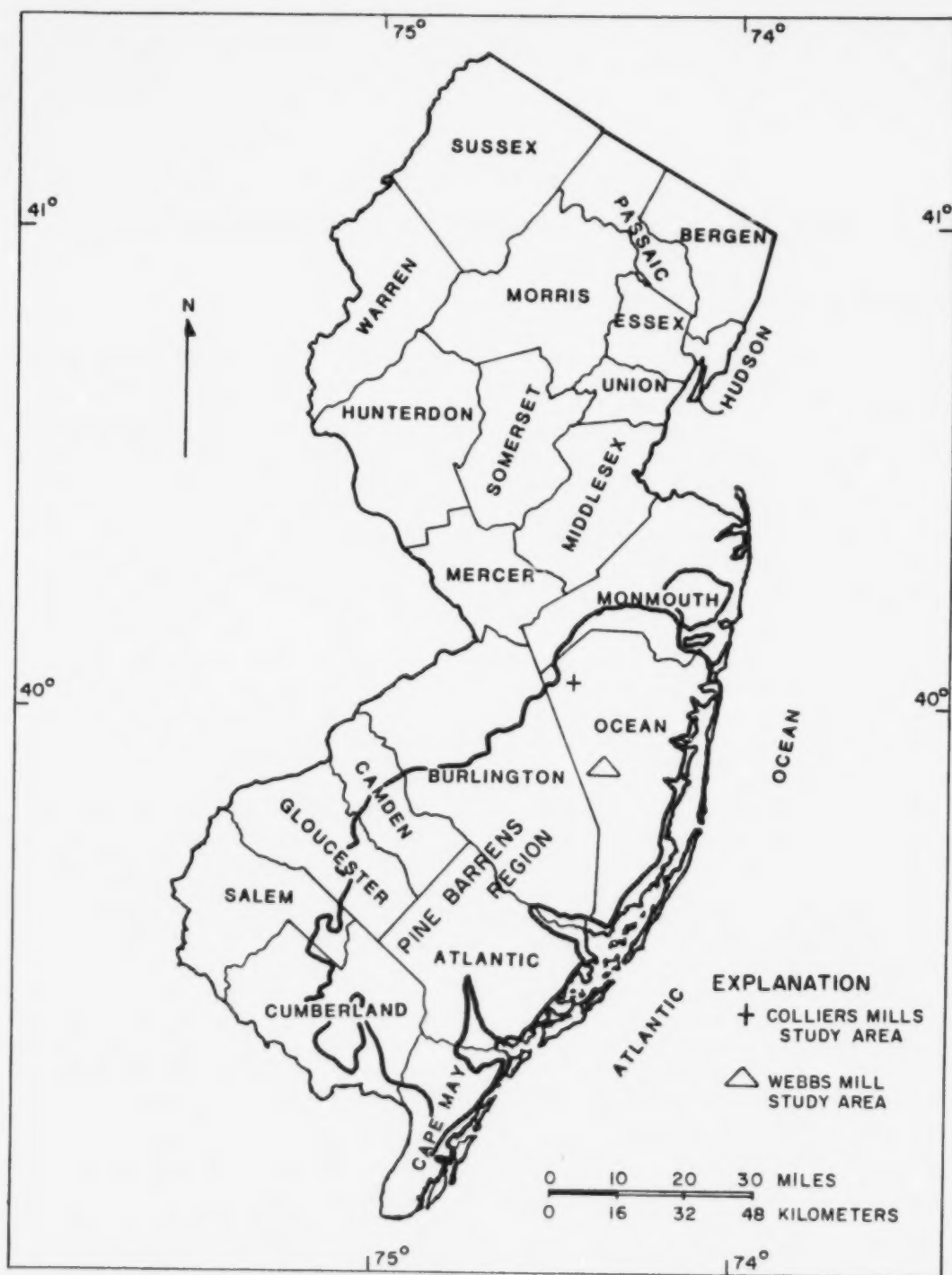


Figure 1. Map of New Jersey showing location of study areas.

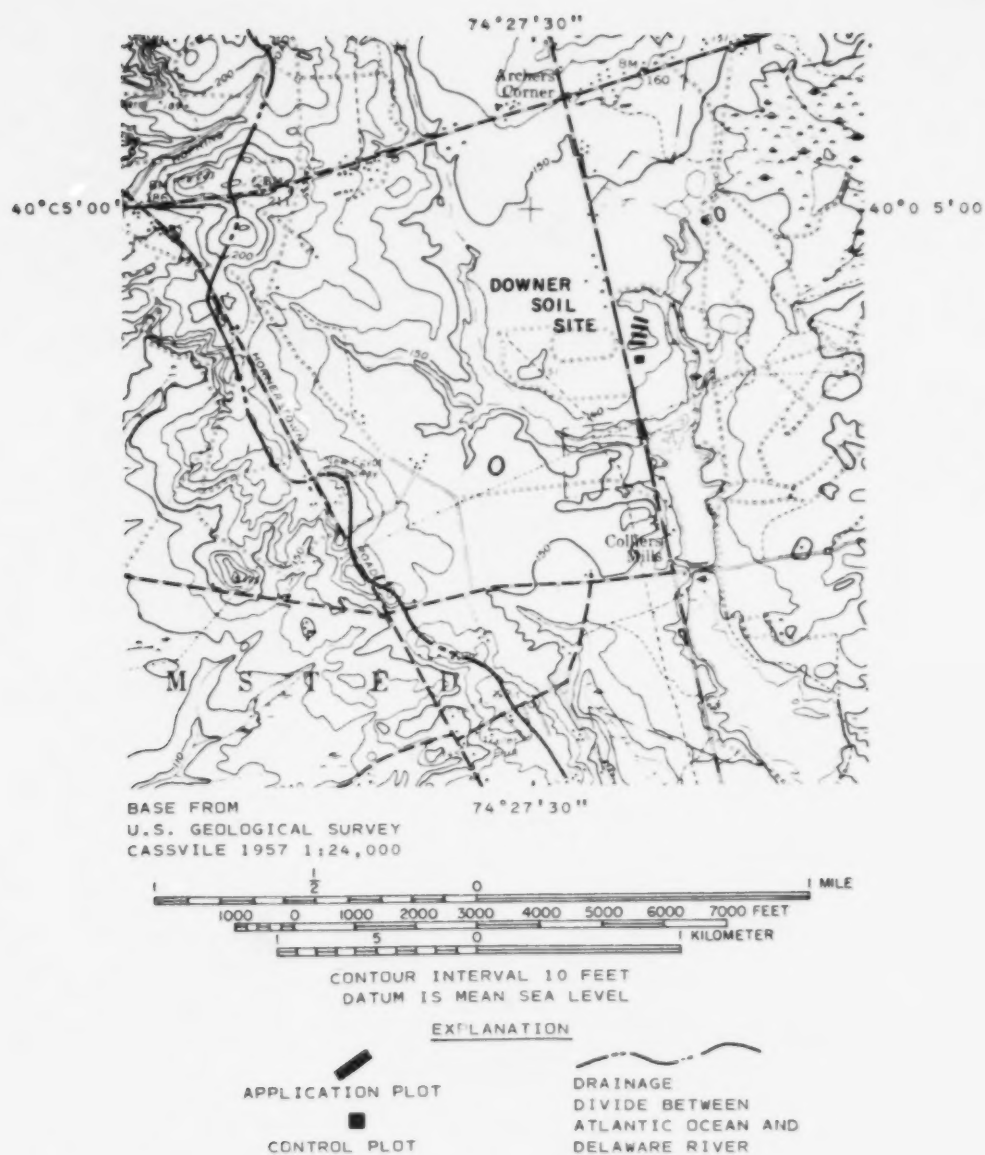
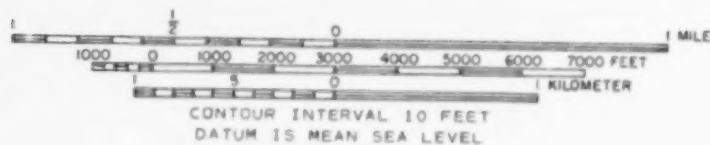


Figure 2. Location of Colliers Mills, Downer Site 1.



BASE FROM U.S. GEOLOGICAL SURVEY
WHITING 1957, WOODMANSIE 1957,
KESWICK GROVE 1957, AND
BROOKSVILLE 1957 1:24,000



EXPLANATION

	APPLICATION PLOT		DRAINAGE DIVIDE BETWEEN ATLANTIC OCEAN AND DELAWARE RIVER
	CONTROL PLOT		

Figure 3. Location of Webbs Mill, Lakewood Soil Site 1, and Woodmansie Soil Site 2.

New York to North Carolina. At the Colliers Mills site, the soil type belongs to the Downer Series. At the Webbs Mill sites the soils belong to the Lakewood and Woodmansie series. In general, the unconsolidated sedimentary formations beneath the study areas ranges in age from Cretaceous to Holocene and compose a sedimentary column several thousand feet thick. These thick deposits consist of an alternating sequence of relatively permeable beds of sand and gravel and less permeable layers of clay and silt (confining beds). This study, however, is concerned chiefly with the thin surficial sedimentary rock unit named the Cohansey Sand. The Cohansey Sand has been described in Ocean County as a characteristically yellowish-brown, unfossiliferous, cross-stratified pebbly, illmenitic, fine-to very coarse-grained quartz sand that is locally cemented with iron oxide. White, dark gray, and red kaolinitic clays are interbedded with the sands as laminae or lenses that may be as much as 9.1 m (30 ft) thick. The overall thickness within the county ranges from a feather edge along its western margin to 61 m (200 ft) in the southeastern part of Ocean County.

At the study sites, the Cohansey Sand is the surface formation. The thickness at the Colliers Mills site is about 15.2 m (50 ft). The thickness is also the depth to the base of the unconfined aquifer. At the Webbs Mill sites the base of the unconfined aquifer or the top of the first significant confining layer is at a depth of about 28.6 m (94 ft).

Data from more than 120 holes, drilled, augered and driven, during this study indicate that the Cohansey Sand at these sites is heterogeneous and the lithologic continuity of individual layers is not traceable.

SOIL SERIES

Three soils, typical of Ocean County, New Jersey and the coastal plains of the Mid-Atlantic states were selected for sludge disposal studies. Soil of the Downer series represents the best soil (for crop production) available in appreciable quantities in the region under consideration. The Lakewood series represents the most widespread soil type in this region. The Woodmansie is identical to the Lakewood series in A horizon but differs in having a clay development in the B horizon (Figs.4-6).

Specific descriptions are as follows:

Downer Series

The Downer series consists of well drained moderately sandy soils. They form generally under a hardwood forest composed of mostly oaks and hickories.

A typical Downer profile consists of 35.6 to 45.7 cm of sandy loam or loamy sand surface soil. The subsoil is 25.4 to 45.7 cm of sandy loam. Clay content normally ranges from 10 to 15 percent and silt content ranges from 5 to 25 percent. The substratum from 76.2 to 152.4 cm is loamy sand. Part of the substratum in places is gravelly loamy sand. The gravel is dominantly rounded quartzose ranging up to 2.5 cm in diameter.

Downer soils are extremely acid to very strongly acid throughout the profile, unless they have been limed. Permeability is moderate or moderately rapid ranging from 1.5 to 15.2 cm per hour. The available water capacity is moderate based on a depth to 152.4 cm though crop roots are not extensive below the subsoil.

Organic matter content is generally low in most places. Cation exchange capacity for Downer soils normally ranges from 4.5 to 6.0 in the surface soil, 2.0 to 5.0 in the subsoil, and 1.0 to 3.0 in the substratum.

Lakewood Series

Lakewood series consists of excessively drained very sandy soils. They normally form under a coniferous forest composed of mostly pitch or shortleaf pines.

A typical Lakewood profile has a bleached light gray sand surface soil 20.3 cm thick. The subsoil is from 20.3 to 63.5 cm yellowish brown sand. The upper 2.5 or 5.0 cm in places contains a dark accumulation of organic and iron compounds. The substratum from 63.5 to 152.4 cm is sand. In parts of the research plots there was 3 to 4 percent of clay accumulation in the subsoil over the content in the surface soil.

Lakewood soils are extremely acid or very strongly acid unless they have been limed. Permeability is rapid exceeding 15.5 cm per hours. The available water capacity is low. Organic matter content is very low. The cation exchange capacity of Lakewood soils ranges from 0.2 to 1.0 in the surface soil, 0.5 to 1.0 in the subsoil, and 0.2 to 1.0 in the substratum.

Woodmansie Series

Woodmansie series consists of well drained moderately sandy soils with a bleached gray surface soil that exceeds 17.8 cm in thickness. The soils normally form under a pitch pine forest.

A typical Woodmansie profile has a light gray sand surface soil 20.3 cm thick. The subsoil from 20.3 to 76.2 cm is composed of sand in the upper one-third portion and sandy loam in the lower two-thirds portion. The substratum from 76.2 to

152.4 cm is stratified layers of sand and sandy loam in places containing 10 to 30 percent rounded quartzose gravel.

Woodmansie soils are extremely or very strongly acid. Permeability is moderate or moderately rapid ranging from 1.5 to 15.2 cm per hour. The controlling horizon is the subsoil. The available water capacity to 101.6 cm is moderate or moderately low. Organic matter content is low. The cation exchange capacity is 0.5 to 1.0 in the surface soil, 1.5 to 9.5 in the subsoil, and 0.1 to 3.0 in the substratum.

HYDROLOGY

The Cohansey Sand is part of an interdependent stream-aquifer system in the New Jersey Coastal Plain. This deposit constitutes the most widespread and permeable section of saturated sand in the water-table aquifer throughout most of southern New Jersey and receives all of its recharge directly from precipitation. The infiltration capacity is high so that overland flow is of short duration. Most of the soils are capable of accepting more than 160 cm (6.3 in) of water per hour. Thus there is little likelihood that the Cohansey rejects recharge from precipitation except where the water table is at or near the surface. The porosity of the material is in the range of 30 to 40 percent and the weighted average specific yield of samples is in the order of 23 percent.

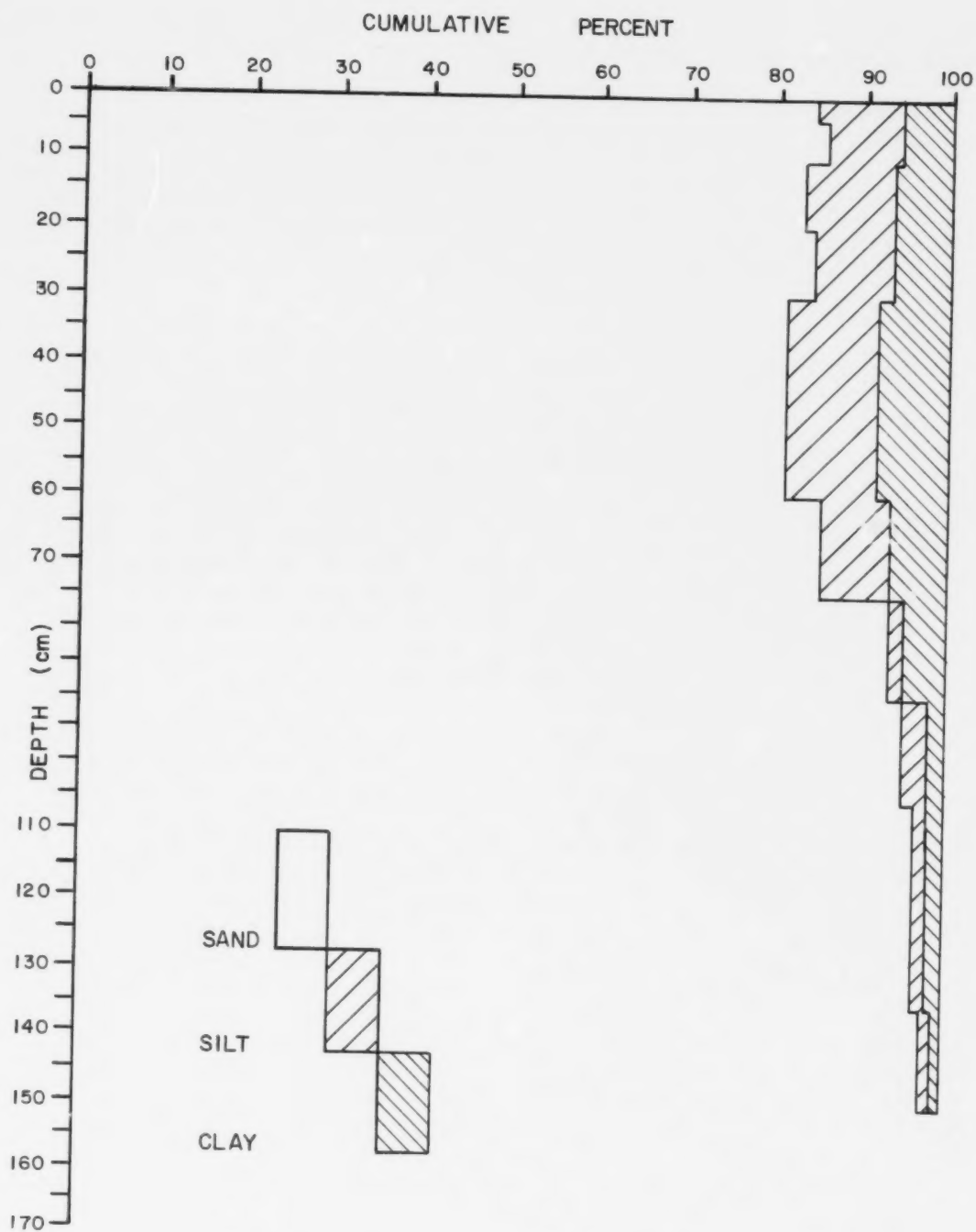


Figure 4. Particle size distribution in the profile of Downer Loamy Sand.

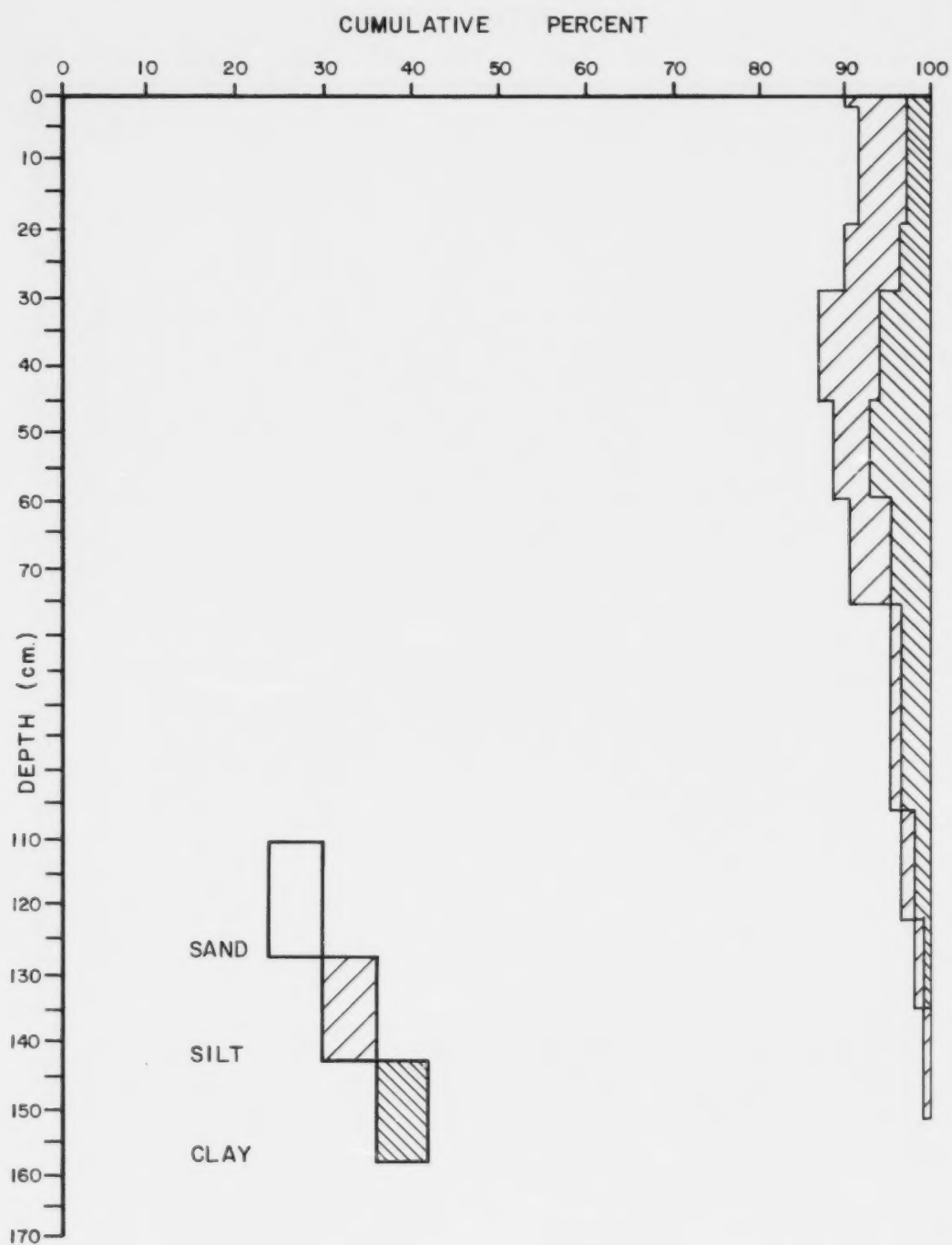


Figure 5. Particle size distribution in the profile of Lakewood Sand.

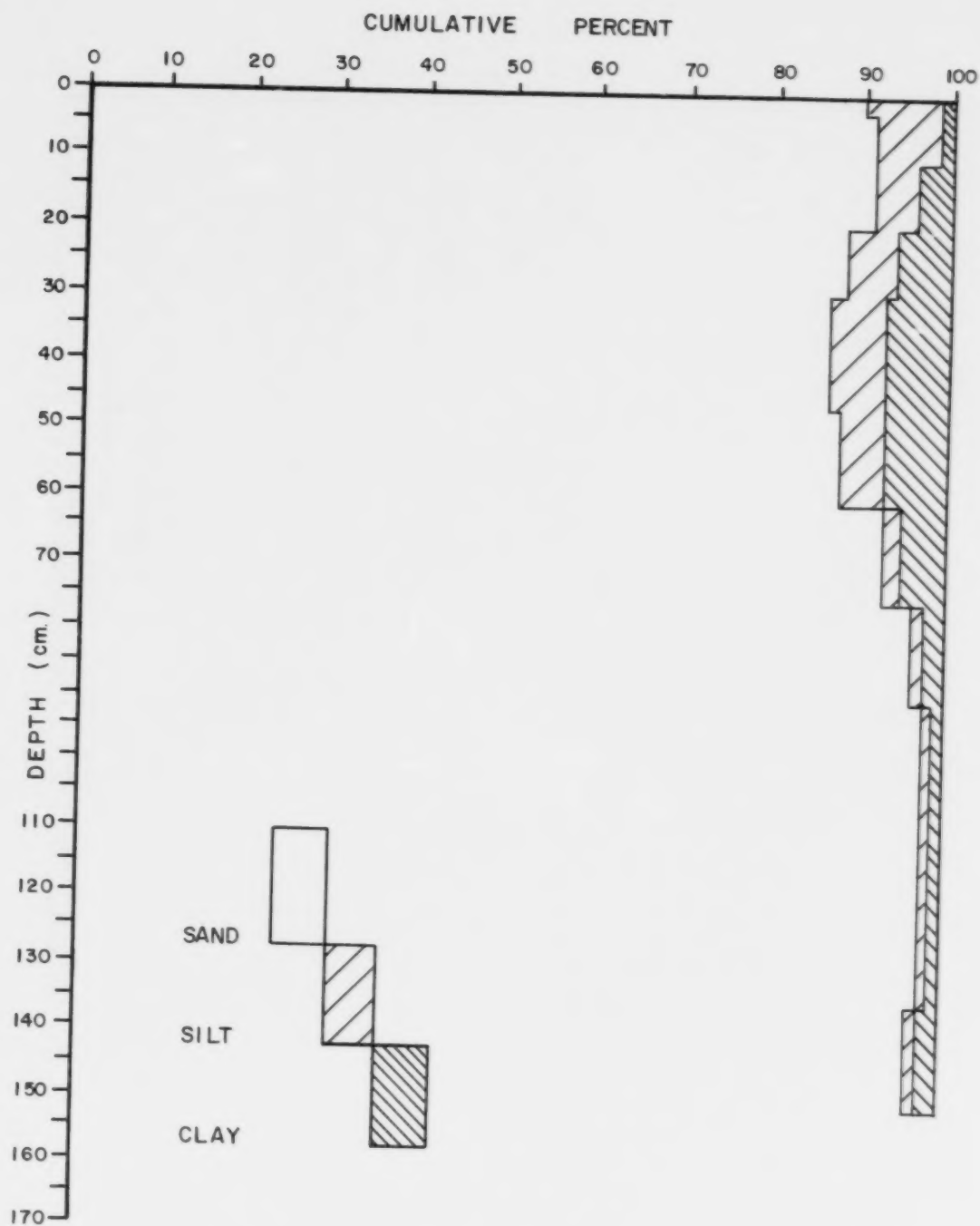


Figure 6. Particle size distribution in the profile of Woodmansie Sand.

SECTION 5

APPLICATION SITES

NATIVE VEGETATION

The sites chosen for the land application of wastewater solids were forested with vegetation typical of these soils. The Downer loamy sand was dominated by white, black, and red oaks (*Quercus alba*, *Q. velutina*, and *Q. rubra*) with an understory of mountain laurel (*Kalmia latifolia*, huckleberry (*Gaylussacia* spp., and blueberry (*Vaccinium* spp.)). The Lakewood and Woodmansie sands were both dominated by small red and black oaks and pitch pines (*Pinus rigida*). These soils also had an understory of huckleberry (*Gaylussacia* spp.) making this the one form of vegetation common to the forested (native vegetation) plots of the three soils of this study. A more definitive description of vegetation of the pine barrens may be found in Harlow and Harrar, 1968.

SITE SELECTION AND PLOT LAYOUT

A minimum number of five plots per site was established as a project requirement. The size of each plot was also established at 929m² (10,000 ft²). The dimension, spacing, and orientation of the plots at each site were determined on the basis of the probable direction and an estimate of the velocity of groundwater flow and the probable path of included contaminants that may occur under each plot.

Plot dimensions of 15.2 x 61 m (50 x 200 ft) were determined to be reasonably adequate. The long dimension of each plot was set parallel to the direction of groundwater flow. The spacing between plots was set at about 30.5 m (100 ft) and the plots were oriented so that the convergence of the groundwater flow paths under the downgradient from the plots would be eliminated or held to a minimum.

A bulldozer with raker teeth was used to clear access roads to selected plot areas. Perimeters of plots to remain in native vegetation were bulldozed to permit traffic by the applicator tractor and trailer. Existing vegetation was cleared from three quarter-acre (0.093 ha = 15.24 m x 60.96 m) plots of each soil series on which grass was to be established. Although an

attempt was made to minimize removal of surface soil, while removing stumps and roots, most of the mull and forest litter and approximately 5 to 10 cm of the surface mineral soil were removed.

Some small stumps and large roots still remaining had to be grubbed and removed by hand to facilitate machine operations. The soil was then disked and dragged in several directions in an attempt to produce a more uniform surface.

Half of the lime required to neutralize soil acidity was applied as hydrated dolomite (45% CaO plus 30% MgO equivalent). On the Downer soil this was 1681 kg/ha total oxides of Ca and Mg, and on the Woodmansie and Lakewood soils, 14101 kg/ha total oxides. This first half was disked into the surface 10 cm. Approximately 50% of the first year's quota of wastewater solids was applied as slurry on the designated 22.4 t/ha/y plots, and as sand bed filter dried cake on the designated 44.8 and 89.6 t/ha/y plots. This was done in an attempt to increase the moisture holding capacity in the plow layer of these droughty soils and enhance the chances of successful establishment of the Midland bermudagrass. After plowing to a mean depth of 20 cm with a 71 cm disk-plow the remainder of the required lime was applied and disked in as before. Cross disking smoothed the field somewhat to permit marking rows on 76 cm centers.

CROP ESTABLISHMENT

The decision to use Midland bermudagrass as the chief vegetative cover for the sites to receive the sewage sludge was based on experiences locally and elsewhere as reported in the literature. It appeared that a vegetative cover should be as stable and uniform as possible during the course of the demonstration. A perennial grass, once established, would minimize land preparation (therefore requiring less management, equipment, and energy) and erosion potential. The grass should also be adapted to sandy soils under consideration and tolerate extremes of fertility rates. Local experiences indicated that Midland bermudagrass would meet these criteria. This grass also has produced higher yields than any other perennial forage ever reported in the State of New Jersey. Bermudagrass has a relatively deep root system and utilizes a great amount of nitrogen.

Midland bermudagrass has been successfully double-cropped with rye. The drilling of rye into the fall-dormant bermudagrass permitted a cool season crop to be taken from the same area of land without jeopardizing the perennial bermudagrass. Thus plant uptake of potential pollutants could be maximized during both cool and warm season without tillage of its associated hazards and expenditures. With multiple surface applications of sludge, loss of nitrogen to the atmosphere would be enhanced and the potential for groundwater pollution would

be reduced.

Plot planting fresh Midland bermudagrass sprigs every 76 cm on ridges left by each tractor-tire-made row was initiated on July 11, 1973, and continued for two weeks. At elevated soil temperatures fresh planting material failed to the extent that between 30-70% of certain plots had to be replanted. On the heavier Downer soil, planted during rainy weather, no replanting was necessary.

Plots measuring 3m x 9m were limed and planted in a similar fashion adjacent to experimental plots. No sludge was applied to these plots. Inorganic fertilizer, 5-4.4-8.3 (N-P-K) at 650.5 kg/ha was applied prior to planting and again the following spring to sustain the Midland bermudagrass to serve as check plots.

LOADING RATES

To assess the effect of rate of sludge application on the environment, solids loading rates of 22.4 t/ha/y, 44.8 t/ha/y, and 89.6 t/ha/y were applied on the three cleared plots comprising three soil types. A 44.8 t/ha/y loading rate was employed on the "native vegetation plot".

WELLS AND WELL CODINGS

Each plot and well were coded in order to computerize all data. The letters CM are used to denote the Colliers Mills site while WM are used for the two Webbs Mill sites. Two digits are placed after the letters. The first denotes the site, and the second signifies the particular plot. Webbs Mill has two sites, therefore, the digits 1 and 2 were used to denote the Lakewood and Woodmansie sands respectively, while the following digit denotes plots 1, 2, 3, 4, or 5. For the Colliers Mills site, only the number 1 was used because there was only one type of soil at this location. The letters A, B, C, D, and E were used to code the wells and profile samplers on or down-gradient from each plot. For example, the following three (3) coded wells or profile samplers can be translated as:

WM12A: Webbs Mill, Lakewood site, plot 2, Well A
WM24D: Webbs Mill, Woodmansie site, plot 4, Well
or profile sample D₁
CM13C: Colliers Mills, Downer site, plot 3, Well
or profile sample C₂

(Subscripts identify the sampling zone of the profile sampler)

The plot layout, distribution of the wells and equipment and assigned plot loading rates are shown in Figures 7, 8, and 9.

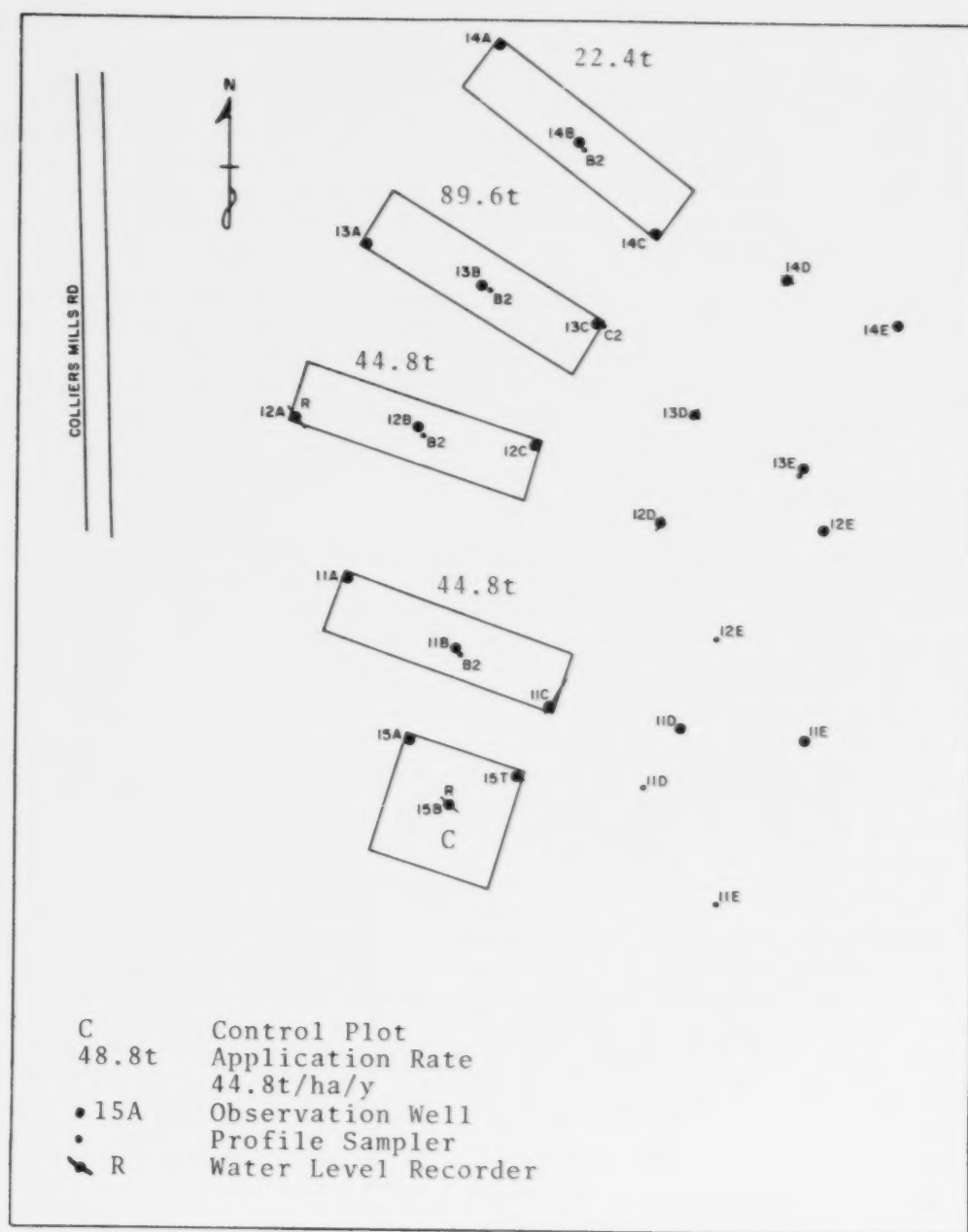


Figure 7. Colliers Mills (Downer Soil) Site showing distribution of wells and equipment in or near the plots.

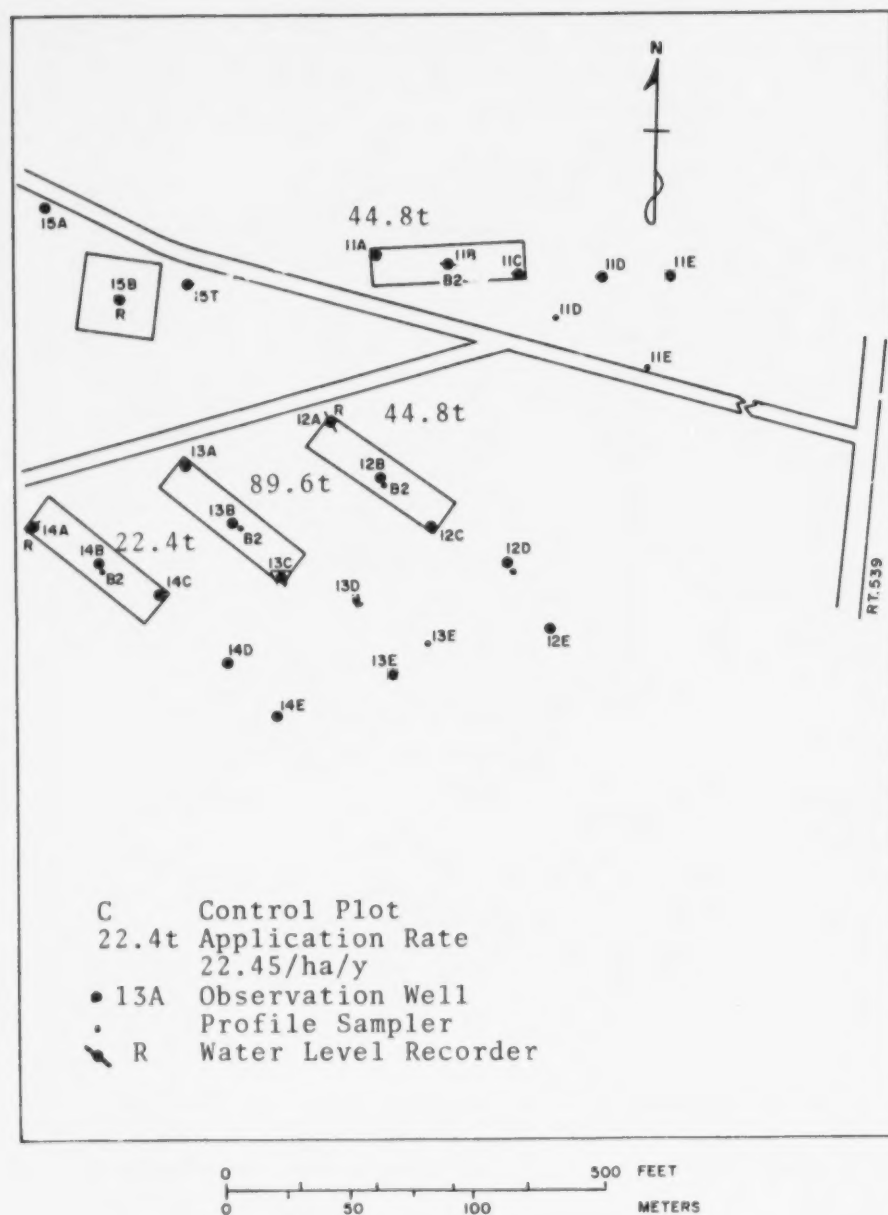


Figure 8. Webb's Mill (Lakewood Soil) Site 1 showing distribution of wells and equipment in or near the plots.

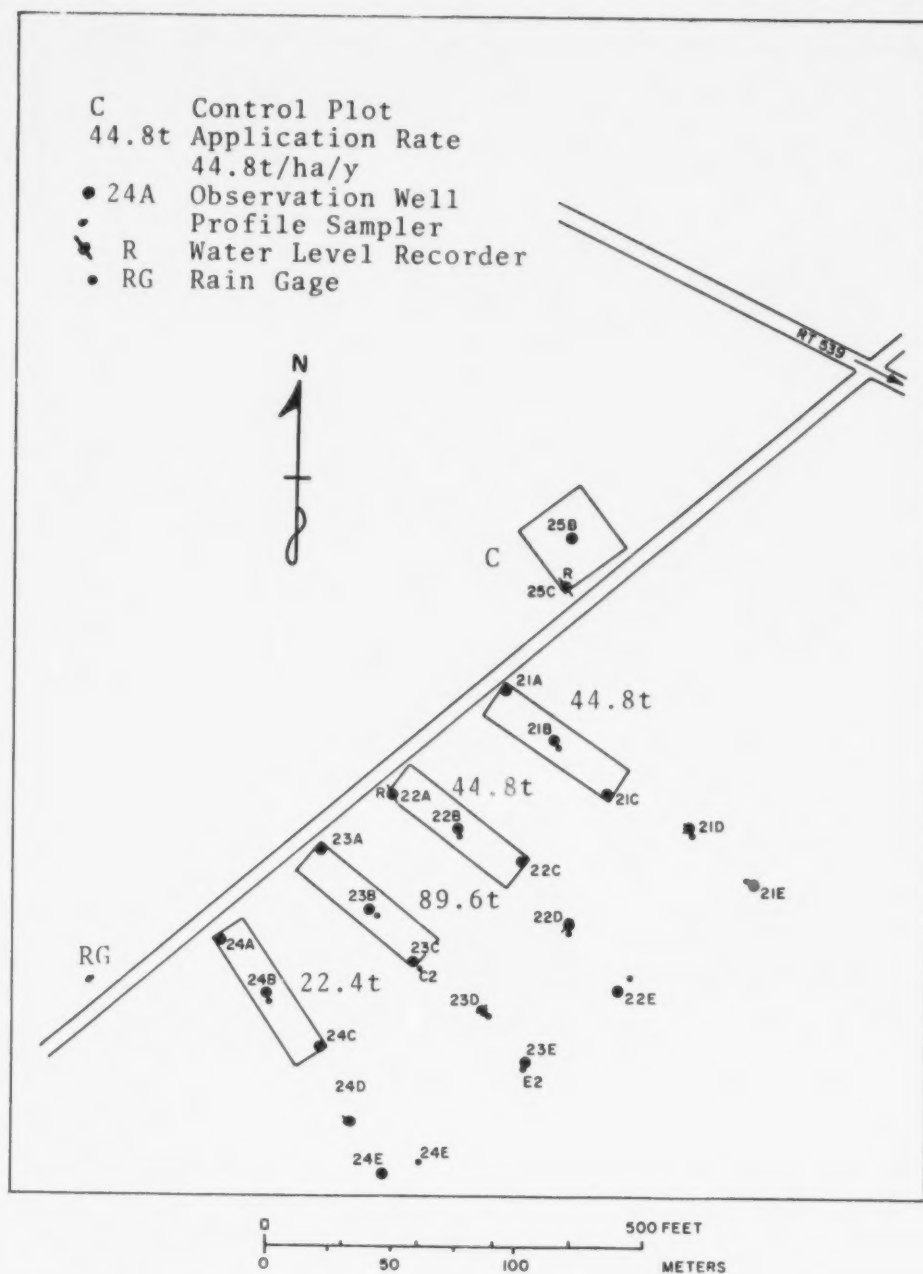


Figure 9. Webb's Mill (Woodmansie Soil) Site 2, showing distribution of wells and equipment in or near the plots.

GROUNDWATER MONITORING

Groundwater monitoring procedures were varied depending upon the specific condition being studied. In general, samples were collected after bailing or pumping a well at least long enough to evacuate the equivalent volume of water in the well bore. This approach was used so that a fresh and representative sample of water from the aquifer would be obtained. The problem encountered with this method was that bailing or pumping distorts the flow field and the water moving to the artificial discharge point converges at the point in a three dimensional pattern. Thus, if a contaminated zone was only a small part of the aquifer, and lies entirely within the distorted flow field, a severe dilution effect occurs in the discharge water. This produces a non-representative sample of the water quality in transit within the aquifer, particularly if the aquifer screened contains stratified layers of varying permeability and stratified layers of contaminants.

In this demonstration study, the collection of "representative" water samples of the aquifer was needed only to define the background water quality prior to the application of the sludge. Thereafter, the collection of samples was based on the need to know when and where any contaminants reached the upper part of the saturated zone under the treated plots, and the subsequent migration both vertically and horizontally. To obtain the least disturbance to the flow field extractions of only the quantity of water needed for laboratory analyses, through a system of collection using a "skim" sampler, a "profile" sampler and in places specific conductance monitors were utilized.

Skim Sampling

The device used for "skim" sampling is shown in Figure 10. The sampler was lowered slowly down the well into the saturated zone so as not to seriously disturb the relatively static condition of the natural flow field about the well. The sampler was submerged to the inlet holes and the water was permitted to flow into the sampler from just below the water level surface. The sampler is then withdrawn and the sample is poured into standard containers.

Profile Sampling

The profile samplers consisted of a series of tubes spaced in sealed multiple sand or pea gravel-packed compartments in a screened well point which was driven to specific depths. This installation permits the simultaneous collection of a number of groundwater samples by a vacuum manifold from predetermined depths in a single hole. (Figure 11). Sampling in this manner permits the determination of the approximate thickness of any

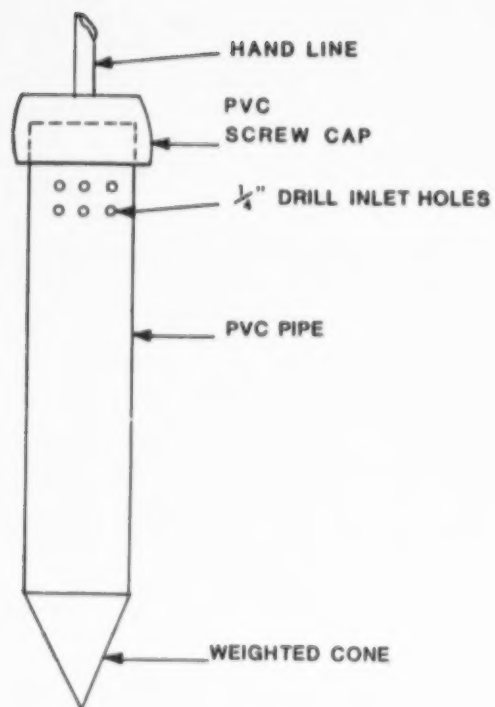


Figure 10. Diagram of Skim Sampler.

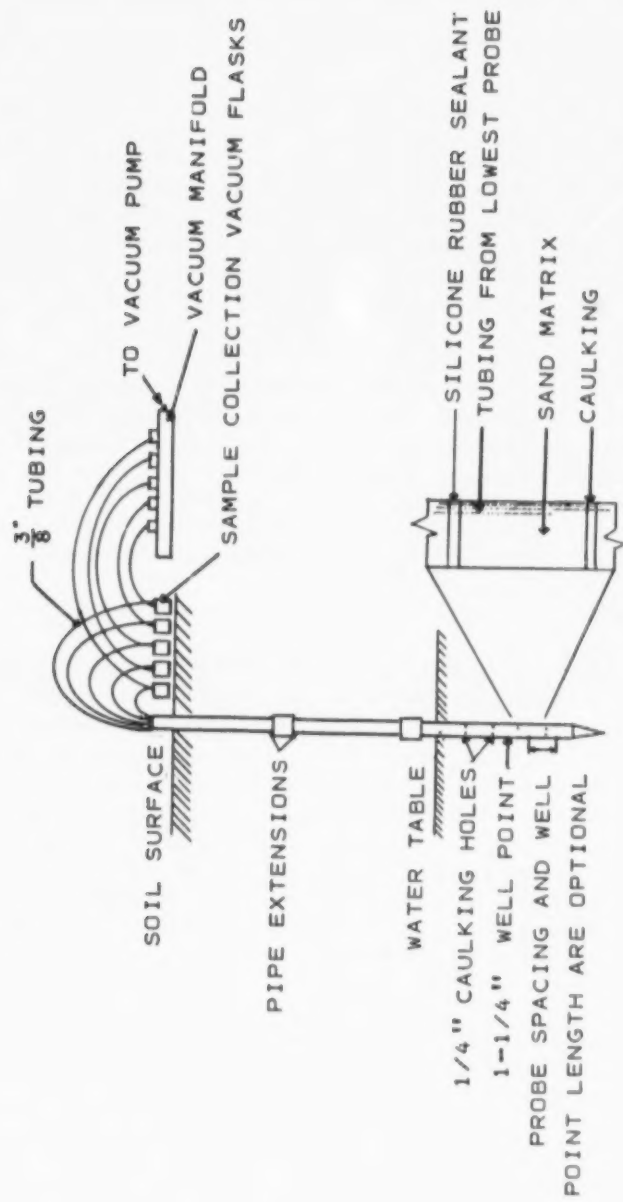


Figure 11. General construction details of the Ground-water Profile Sampler. Modified from Hansen and Harris, 1974.

contaminated zone and the transient depth of maximum concentration. Such profiles collected at each of the plots adjacent to the wells and downgradient from the plots indicate the progress of contaminant movement and changes in concentration.

GROUNDWATER TABLE FLUCTUATIONS

Changes in aquifer storage as a result of recharge from precipitation and discharge into surface streams are manifested in the fluctuation of the water levels. The water level in the Cohansey Sand is constantly changing in response to recharge from precipitation as indicated in Figures 12-15 by a typical hydrograph from an observation well and a precipitation graph in the study area from 1973-1976. This pattern of natural fluctuation can be considered typical for the Cohansey Sand. The high water levels during May, June, and July 1975 reflect above normal precipitation during that summer. The rise of the water levels is in response to the recharge from precipitation. The rise and decline of water levels can be considered as pulses. Therefore, it can be assumed that any stratified concentration of contaminants in the unsaturated zone that lies within the region of water-level fluctuation will enter the groundwater system as pulses thereby producing stratified concentration of contaminants in the groundwater system.

GROUNDWATER MOVEMENT

The direction of groundwater flow in a homogeneous granular porous material is defined by streamlines which are perpendicular to the equipotential lines (lines connecting points of equal head) in an aquifer. Streamlines give the instantaneous flow pattern, which in general changes under non-steady flow conditions. Consequently, the periodic mapping of the groundwater surface is a prerequisite for defining the lateral movement of groundwater and included contaminants. The initial water table maps constructed at the beginning of the project identified the general direction of groundwater movement and the gradient of the surface and thus influenced the orientation and spacing of the plots.

In the study area, water table maps were constructed for each site for different times beginning in April, 1973. These typical surfaces are shown in Figures 16-18.

The contours (equipotential lines) indicate the altitude of the water table and the difference in altitude on the line normal to the equipotential lines indicate the gradients. When a conservative contaminant (chloride, or nitrate) reaches the water table anywhere below the application plots, it will move, under the existing gradient with the groundwater in a definable flow path toward the stream. However, under changing conditions of recharge, discharge, and gradients, the flow path directions

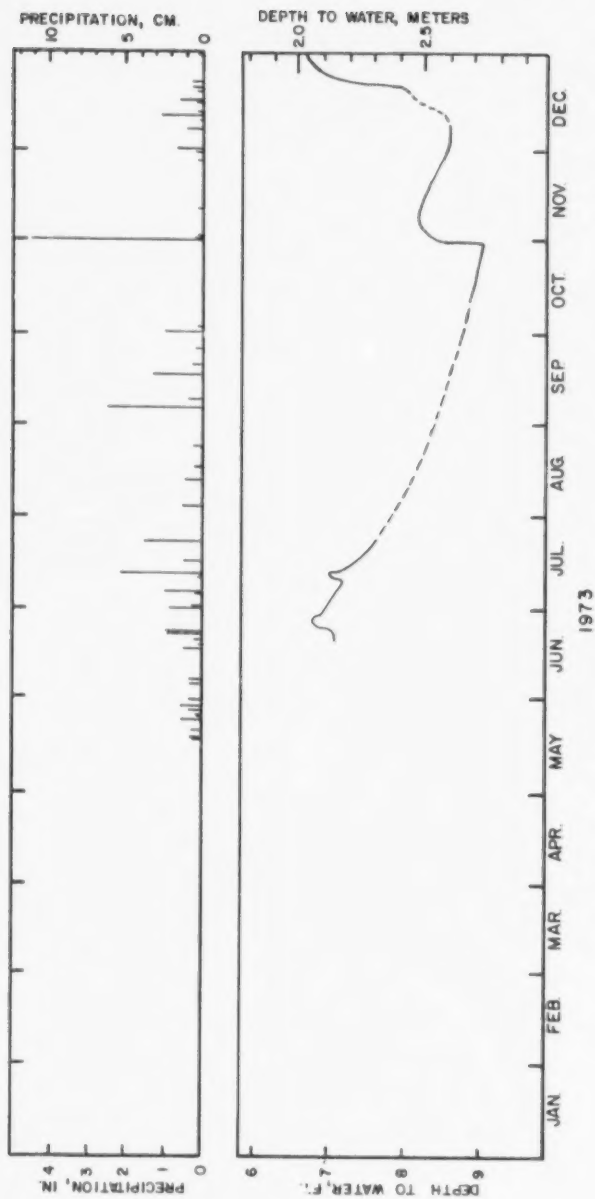


Figure 12. Water level fluctuation in Well CM15A and precipitation data at Colliers Mills Site 1, 1973.

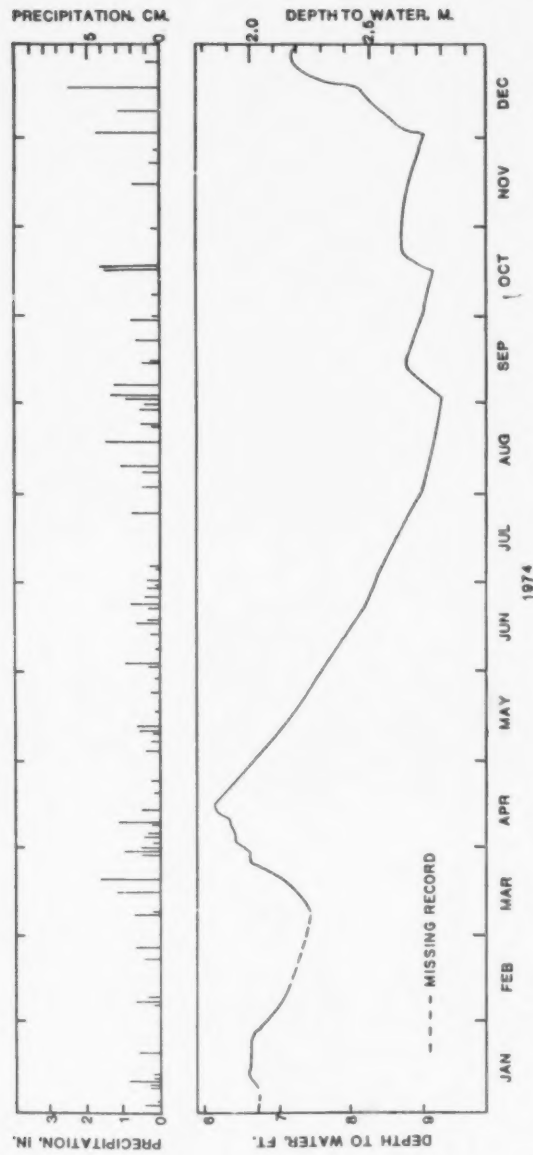


Figure 13. Water level fluctuation in Well CM15A and precipitation data at Colliers Mills Site 1, 1974.

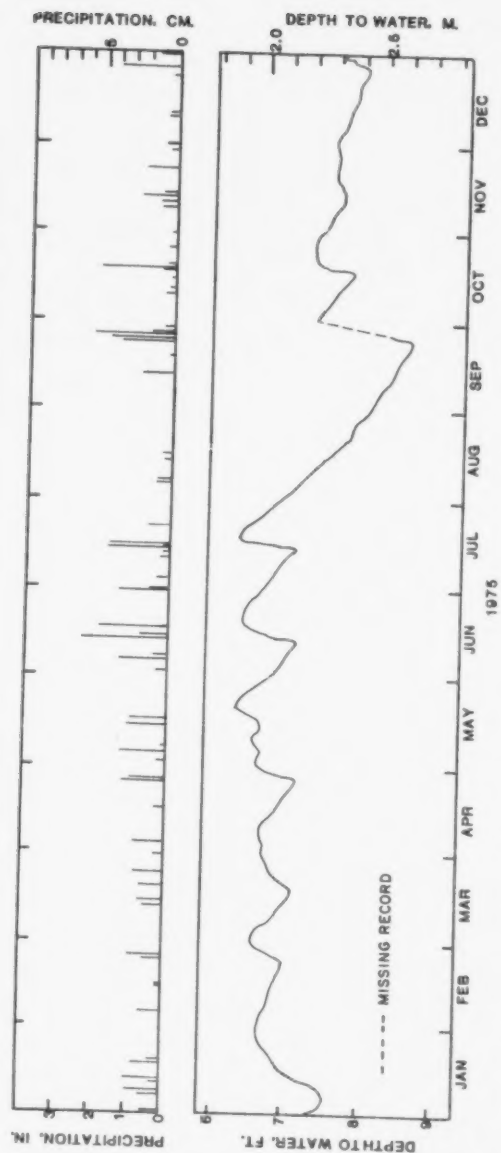


Figure 14. Water level fluctuation in Well CM15A and precipitation data at Colliers Mills Site 1, 1975.

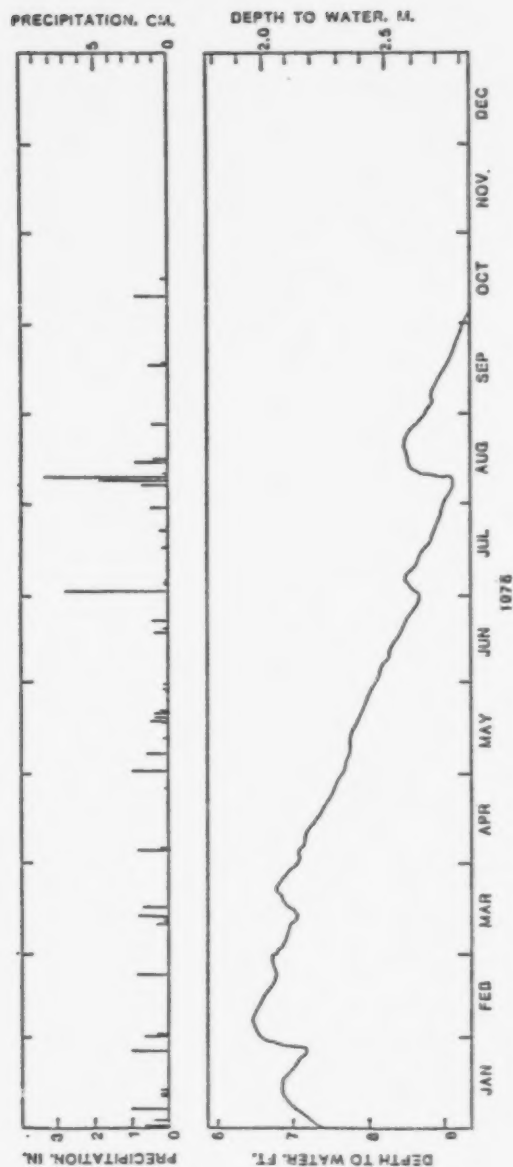
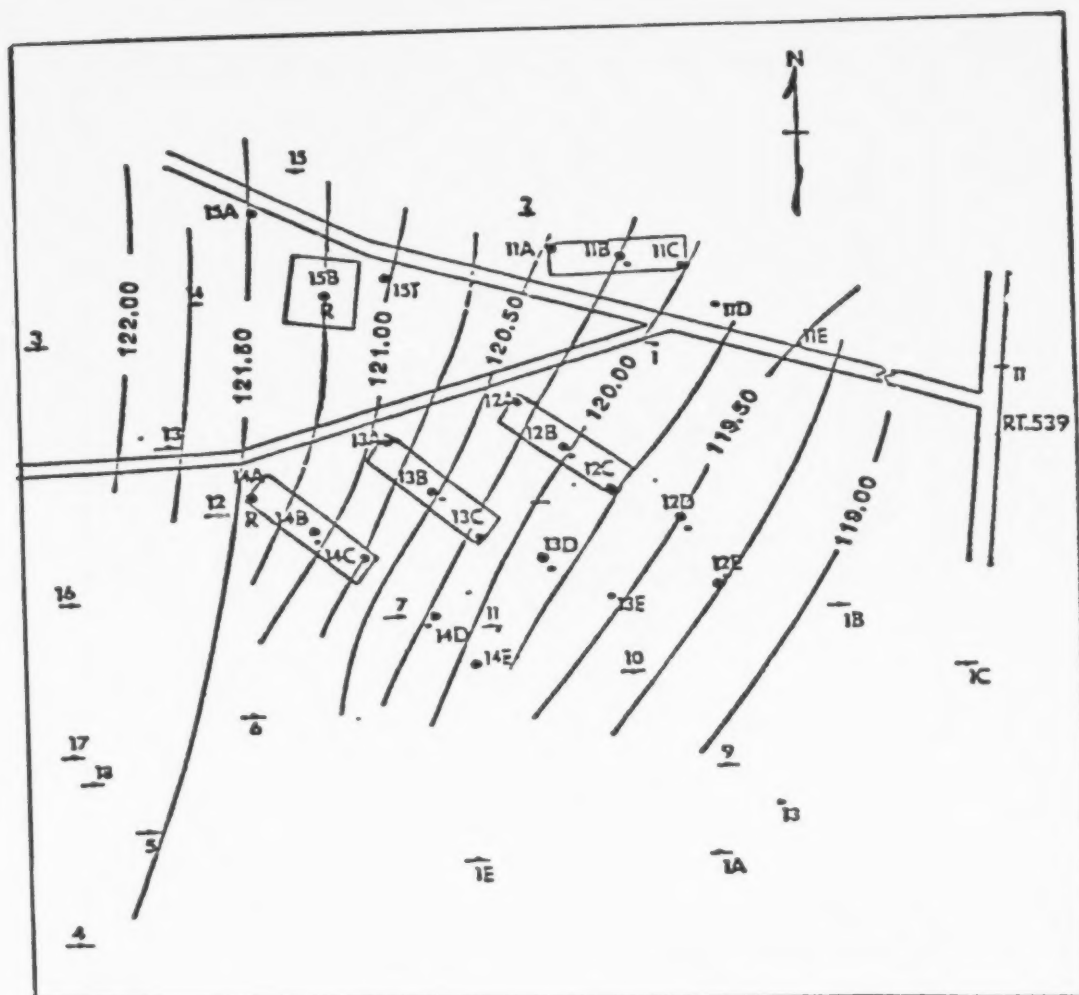


Figure 15. Water level fluctuation in Well CM15A and precipitation data at Colliers Mills Site 1, 1976.



• 13A OBS Well

— Auger Hole

• Sampler

• R Water Level Recorder

~ 120.00 ~

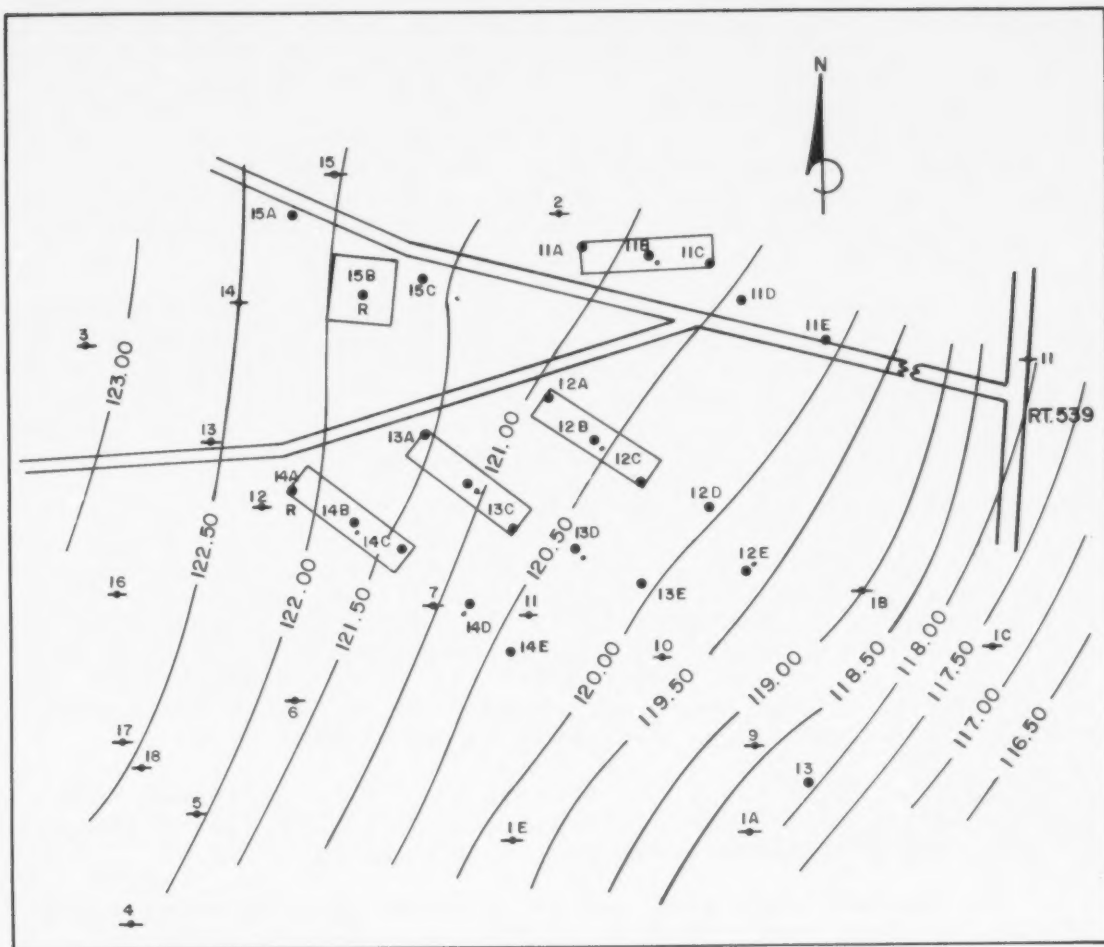
Water Level Contour

Conversion Factor: 1 foot - 0.3048 Meter

0 50 100 500 FEET
0 50 100 METER

Lakewood Site - Webbs Mill

Figure 17. Water Table Map at Webbs Mill Site 1, July 9, 1974.



- 13A OBS WELL
- AUGER HOLE
- SAMPLER
- R WATER LEVEL RECORDER
- 120.00 WATER LEVEL CONTOUR
- CONVERSION FACTOR: 1 FOOT = 0.3048 METER

LAKWOOD SITE WEBBS MILL

Figure 18. Water Table Map at Webbs Mill Site 1,
April 5, 1976.

also change thus causing shifts in the direction of flow. This variability of the natural flow field causes a spreading that is apparently part of the dispersion mechanism. In order to determine the approximate amplitude of the apparent changes in flow directions, streamlines were constructed so that they would pass through the center of each plot for each condition illustrated by the series of water table maps. It was found that the apparent horizontal shift or change in direction of flow along streamlines passing through the center of the plots ranged from 8 to 25 degrees.

In summation the horizontal angular shifts that occur under variable recharge-discharge conditions may be a great factor in dispersion. The identification of the zones of contamination in a non-homogeneous groundwater system can be extremely difficult unless sampling procedures are designed relative to the changing flow directions and vertical lithologic variations.

GROUNDWATER VELOCITY-TRACER TESTS

The use of tracers to determine the direction and velocity of groundwater flow has been described by many investigators and it is acknowledged that there are some serious drawbacks in using artificially injected tracers because of the effects and mixing behavior of the tracers with the fluid-solid system. Nevertheless, tracer tests have been used in making reasonable estimates of groundwater flow velocities in many areas.

In this study tracer tests were used for comparison with other methods of aquifer testing to estimate the flow velocity of included contaminants due to the land application of sludge.

The tracers used were sodium chloride and rhodamine B dye. The arrangement and procedure of the tests were planned to limit the time of travel of the tracer to a relatively short period and distance. This was done to avoid, as much as possible, the effects of changes in recharge-discharge conditions that might occur during the tests. Likewise the method of injection and withdrawal of samples was such that the flow field was disturbed as little as possible.

Three injection points and three sampling points were installed at each of the three sites with the injection points at a radius of 1.8 m (6 ft) upgradient from the sampling points (Figure 19). The holes were oriented in a direction estimated to be parallel to the flow path as determined from the latest water table maps and prevailing gradients. Of the three sodium chloride and rhodamine B dye tests only one of each produced useable results. The unsuccessful tests are attributed to locating the sampling points outside the flow path containing the injected tracers.

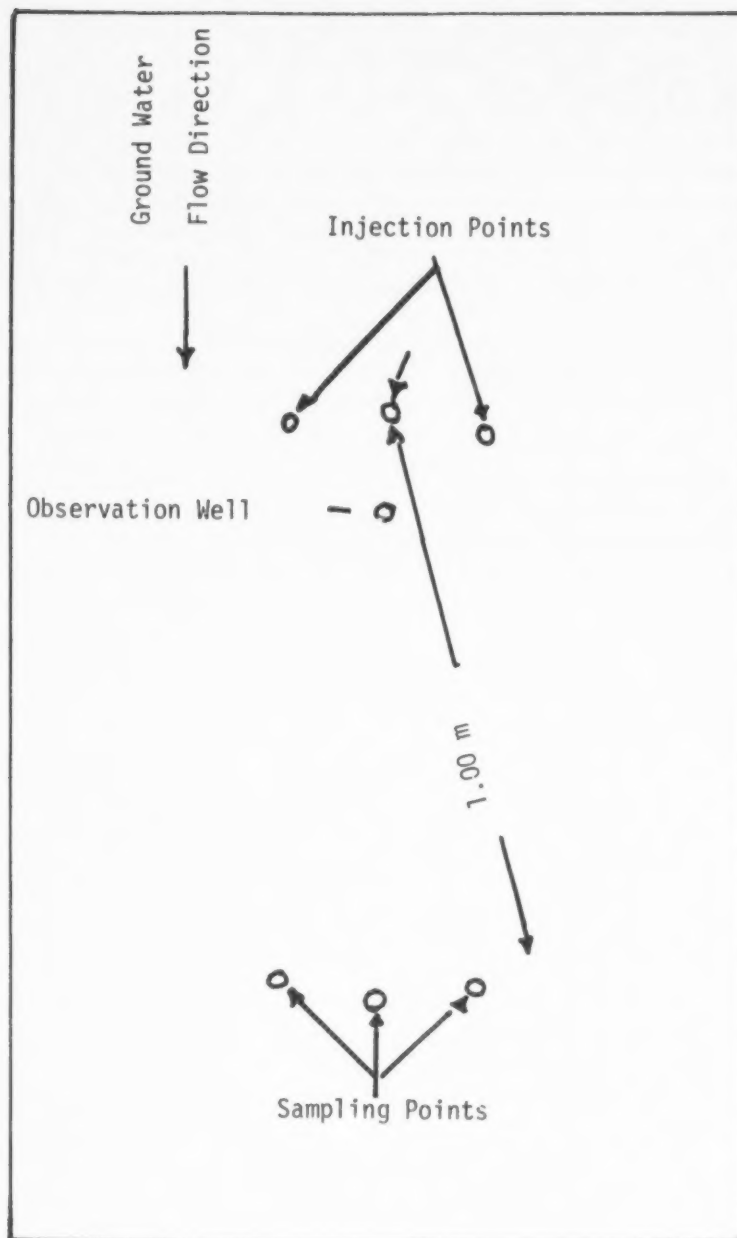


Figure 19. Sketch showing well distribution used in Tracer Test.

In the sodium chloride tracer test three one pound bags of sodium chloride were emplaced at the water table through three (3) augered holes near well CM11E at the Colliers Mills site. The salt bags were crushed with the hand auger to release the salt into the flow path and sampling in the sand points, which were screened 0.6 m (2 ft) into the water table, was continued for a period of 12 days (June 25 - July 8, 1975). The samples were withdrawn from the sampling points by a hand vacuum pump in only such quantities sufficient for analysis. The calculated flow velocity for this test in the shallow part of the aquifer was found to be 0.3 m/g (1.1 ft/day). This velocity compares with 0.47 m (1.5 ft/day) calculated by empirical means.

The same well sampling arrangement was used for the rhodamine B dye tracer test near well WM25B at Webbs Mill site as was used for the chloride test near well CM11E. The same sampling procedure was followed for a period of 45 days (September 30 - November 14, 1975) and the determination of the concentration was by means of a Turner model 111 fluorometer in the laboratory. The calculated longitudinal velocity in the shallow part of the aquifer was about 0.19 m/d (0.62 ft/d). This velocity compares with 0.20 m/d (0.69 ft/d) calculated by empirical means.

SECTION 6

WASTEWATER SOLIDS APPLICATIONS

SOURCE

The type of wastewater solids utilized in this demonstration project was anaerobically digested sludge. Digested sludge results from the anaerobic microbial breakdown of fresh domestic organic matter collected from primary sedimentation and of waste sludge withdrawn from the secondary sewage treatment process.

At the inception of the study in May 1973, wastewater solids were obtained from the South Lakewood Sewer Company secondary treatment facility. Although it was the intention of the investigation to utilize only liquid sludge (slurry), two forms of wastewater solids were trucked from this facility: a dried cake from sludge drying beds and a liquid sludge. In order that the 1973 growing season would not be lost for planting crops and grasses, approximately 40 percent of the annual solids loading had to be applied before planting could commence. Only through the supplemental use of cake could the 40 t/Ha loading be applied in the short period of time remaining in the 1973 growing season.

The liquid sludge trucked from Lakewood immediately brought about an "odor problem" at the application sites. The condition of Lakewood wastewater solids was found to be less than ideal; i.e., the volatile solids content was too high for the sludge to be characterized as a "well stabilized" digested sludge. Analysis of the Lakewood digester showed poor volatile solids destruction, a relatively high volatile acid content for digested sludge, and a pH less than 7.0. As a result of the failing digester, the Neptune Township Sewerage Authority, Neptune, New Jersey, was used as the source of wastewater solids from late July 1973 until completion of three years of solids application ending in October 1975.

CHARACTERISTICS

Analytical Methods and Procedures

All wastewater solids composite samples were analyzed for 21 chemical constituents (Table 5). However, before any of

TABLE 5. CONSTITUENTS FOR WHICH WASTEWATER SOLIDS WERE ANALYZED

pH	Manganese (Mn)
Total Solids (%)	Cadmium (Cd)
Volatile Solids (%)	Lead (Pb)
Ash (%)	Chromium (Cr)
Total Nitrogen (as N)	Copper (Cu)
Total Phosphorus (as P)	Nickel (Ni)
Magnesium (Mg)	Zinc (Zn)
Iron (Fe)	Aluminum (Al)
Sodium (Na)	Mercury (Hg)
Potassium (K)	Boron (B)

the components of the cake and liquid sludge could be determined, each sample had to undergo a six hour acid digestion (macro-Kjeldahl digestion unit). This process is necessary to insure complete solubilization of all inorganic solids constituents. Upon completion of the acid digestion, each sample was then analyzed for its individual components according to the methods and procedures recommended in Standard Methods for the Examination of Water and Wastewater.

Sludge Cake

Cake samples from the two Webbs Mill sites were composited into one sample and a similar composite for Colliers Mills was obtained for characterization. Since all the cake from the same Lakewood drying beds and preliminary tests showed little variability in total solids and total volatile solids content from load to load, a large composite from each site was utilized to obtain the "average" concentration. It should be noted that a large percentage of the cake was composed of sand. A large portion of sand had been removed with the cake when it was mechanically removed from the Lakewood sludge drying beds. Additional sand was accumulated when cake was mechanically transferred from standing piles at the application site to the manure spreader. Therefore, the cake applications had to be corrected for this excess of sand in order to approximate the equivalent amount of cake to liquid sludge on a dry weight basis. This would insure that the proper loading rates of wastewater solids was maintained.

Liquid Sludge

Liquid sludge samples were collected from each tank truck load. It was assumed that the mixing conditions created when loading the tank truck and during transportation to the application site would be sufficient to keep the load homogeneous. Therefore, the sample could be considered "representative".

For a comprehensive analysis, samples from 10 to 20 tanker loads were composited, since constituents of each truck load did not vary much in solids and volatile solids content. Table 6 summarizes the "average" wastewater solids characteristics for the cake, 1973 liquid sludge, 1974 liquid sludge, 1975 liquid sludge and for the three years of solids application.

The towns of Lakewood and Neptune could be characterized as growing communities with primarily "domestic" sewage. The chemical concentrations of sludge constituents encompassed ranges normally observed in the literature for "domestic" sludge with the possible exceptions of relatively high concentrations of iron and total nitrogen. Zinc concentrations were also typically high in a few of the tanker loads analyzed.

TABLE 6.

WASTEWATER SOLIDS CHARACTERISTICS

CHEMICAL CONSTITUENTS	CAKE	1973 SLUDGE	1974 SLUDGE	1975 SLUDGE	3 YEAR* AVERAGE
PH	***	6.8	7.0	7.0	7.0
TOTAL SOLIDS (%)	71.0	5.1	4.3	3.8	4.3
VOLATILE SOLIDS (%d.w.)	20.9	58.1	61.5	61.9	60.8
INERT SOLIDS (%d.w.)	79.1	41.8	38.6	38.1	39.2
-----MILLIGRAMS / KILOGRAM -----					
NITROGEN(N)	8485.0	44063.6	58561.4	55675.6	53942.7
PHOSPHOROUS(P)	4758.7	26520.5	48889.9	31559.4	36891.0
IRON(Fe)	5817.1	31787.4	33520.6	40636.7	35791.5
MANGANESE(MN)	39.4	289.0	312.8	302.8	303.2
MAGNESIUM(MG)	1127.2	3879.5	2634.6	2403.8	2850.1
CALCIUM(CA)	17301.9	56615.5	47893.5	28018.6	42494.6
SODIUM(NA)	413.3	6306.3	6186.5	4079.5	5418.4
POTASSIUM(K)	130.0	1918.6	1439.4	1304.3	1504.9
CADIUM(CD)	10.1	25.2	13.3	17.6	17.9
LEAD(PB)	207.8	443.8	379.5	286.3	359.9
CHROMIUM(CR)	14.7	78.0	51.1	50.5	57.4
COPPER(CU)	289.8	868.9	893.2	962.3	913.4
NICKEL(NI)	8.5	43.7	48.9	36.5	43.0
ZINC(ZN)	247.2	1964.6	2919.6	2828.9	2653.0
ALUMINUM(AL)	3390.6	10090.1	9338.5	9383.9	9538.5
MERCURY(HG)	0.2	2.0	4.0	1.0	2.4
BORON(B)	160.2	378.5	352.2	191.0	297.6

* These figures were calculated by averaging the results of each day's field tests over the 3-year period.

APPLICATION METHODS AND TECHNIQUE

Introduction

The role of Rutgers' Department of Biological and Agricultural Engineering was to design, procure, assemble, and maintain the required field equipment for the application of liquid sewage sludge. Subsequent reassignments added topographic surveying and mapping of the plots, supervision of land clearing and roadwork with the cooperation of the New Jersey Division of Fish, Game and Shellfisheries, transportation of sludge and maintaining records regarding dates and rates of application. Assembly of components, modifications of equipment and major repairs were done in the Agricultural Engineering Shop in New Brunswick, N.J. with the cooperation of the shop foreman and staff.

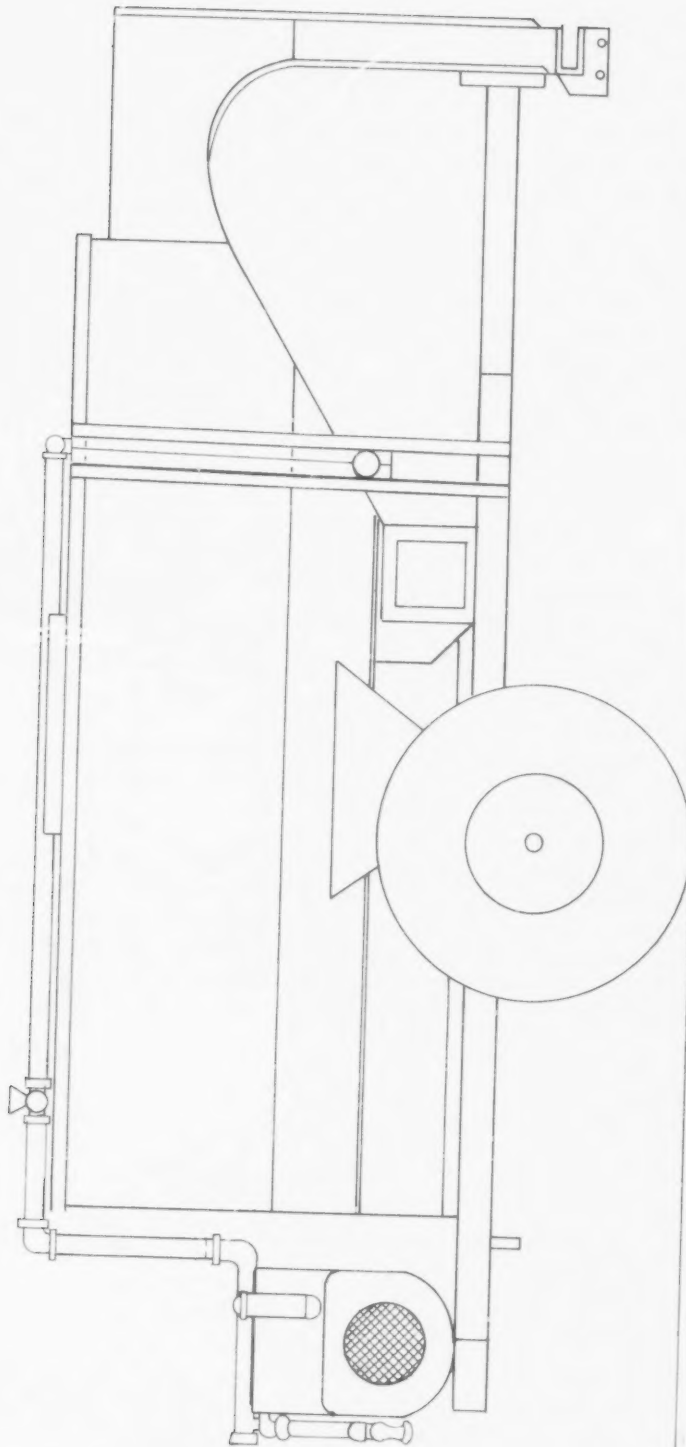
Transportation

The collection, transportation and application of sewage sludge to the Ocean County Sewage Sludge Application Sites were conducted by a three-man crew administered through the Department of Biological and Agricultural Engineering. The Department was responsible for the transportation of sewage sludge approximately 50 miles from the Lakewood and Neptune plants to the sites, transferral of the sludge from the highway tank trailer to the application unit, application of the sludge to the plots and collection of all sewage sludge samples. The equipment was also maintained by the Department and moved six to eight miles daily for security.

The equipment included a 1500 gallon field application unit, an International Harvester 574 farm tractor, a 4-wheel drive International Harvester $1\frac{1}{2}$ ton stake body truck with field maintenance, sampling and sanitation equipment, a single axle International Harvester highway tractor and a 3000-gallon highway tank trailer. Six 3000-gallon loads were transported and applied weekly.

Methods and Techniques

A multipurpose field application unit was designed by the Department of Biological and Agricultural Engineering, assembled by Agway, Inc., and is shown in Figures 20 and 21. It has a capacity of 1500 gallons, a 9-inch ribbon auger and three outlets for discharge of sludge. A Gorman-Rupp Model 203A Pump Kit was mounted on an extended platform at the rear of the applicator. Two 12 ft long irrigation booms were mounted on the trailer tank for spray irrigation to the experimental test plots. A Gorman-Rupp high pressure one-inch nozzle was installed on a two-inch manifold and used to spray-irrigate liquid sewage sludge on natural vegetation plots with trees in excess of eight feet in height. Both the irrigation boom and spray nozzle are presented



SIDE VIEW

Figure 20. Sewage Sludge Application Unit.

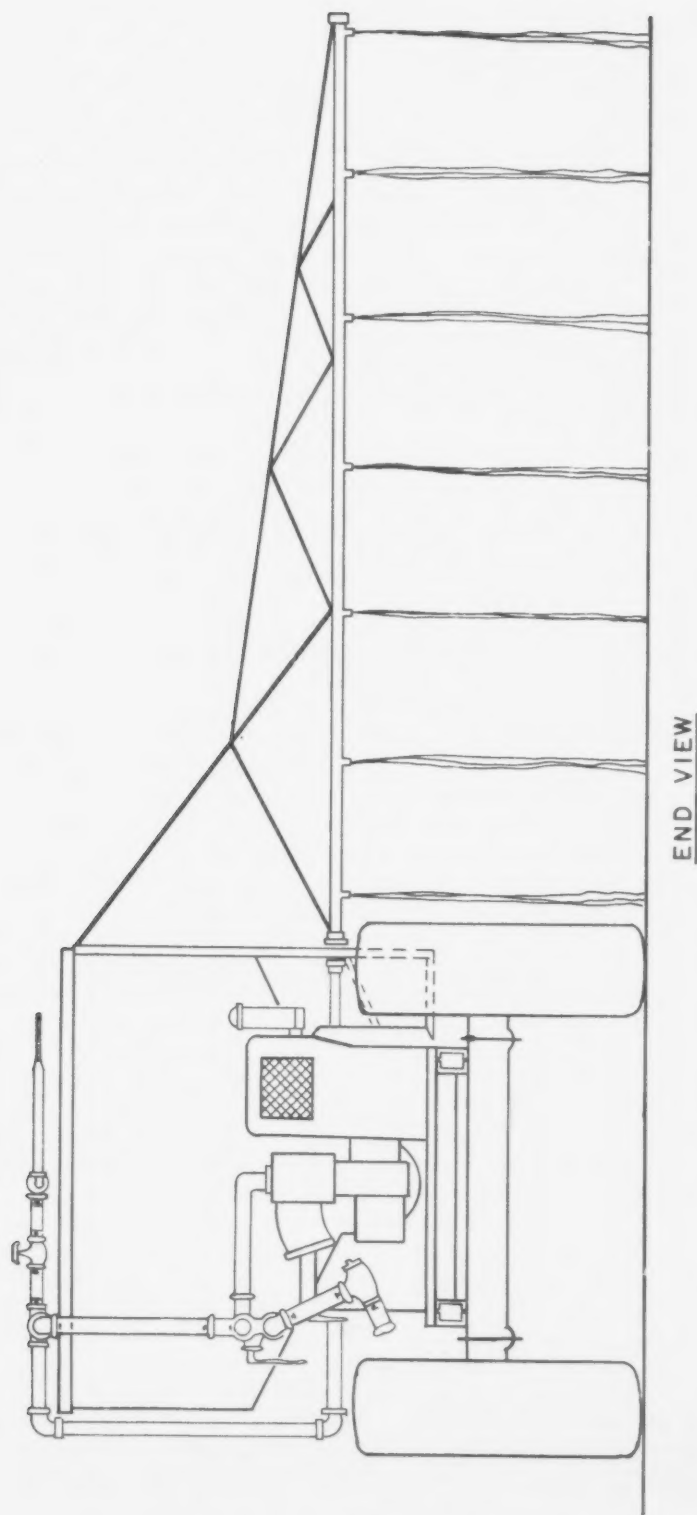


Figure 21. Sewage Sludge Application Unit.

in Figures 20 and 21.

The Sewage Sludge Applicator Unit performed satisfactorily after several modifications to its design. The original unit was not designed to support the additional weight of the pump and pump engine within the gooseneck tongue and required the addition of a new two-wheel chassis. The engine and pump were mounted on the rear of the chassis permitting sludge removal from the rear discharge port. This new chassis design performed satisfactorily and resulted in no further structural problems. The unit was originally designed to use two spray irrigation booms. Because of difficulty in regulating the flow of sludge through each boom and increased problems with maneuverability, one of the booms was subsequently removed.

The techniques of application which were evaluated as part of this project include a high pressure spray boom type distribution system and a high pressure nozzle spray gun. These techniques were utilized to apply liquid sewage sludge to two types of vegetative plots. The simulated spray irrigation applicator was used to apply liquid sludge uniformly to 12 different grass plots at three rates of application. The high pressure gun was used to spray-irrigate three natural vegetation plots.

The simulated spray irrigation boom as shown in Figure 21 consisted of 12 ft long, four-inch diameter irrigation pipe with one-inch diameter spray nozzles spaced at two feet intervals along the boom. The boom was hinged to a frame mounted on the side of the tank. By swinging the boom out over the plots, while regulating the flow of sludge, the operator could apply the liquid sludge as shown in Figure 21. Because of the low solids concentration of the slurry and the design of the boom, several passes over each plot were required to apply the correct amount of sludge. Modifications in the application schedule were sometimes required because of the time required in harvesting the crops, the variable weather conditions, the planting of cool season grasses, equipment maintenance, and availability of sludge.

Natural vegetation sites were spray irrigated with a high pressure nozzle gun. The gun was constructed of a two-inch galvanized pipe reduced to a one-inch standard nozzle. From a service road, the operator directed the spray stream into the natural vegetation plots covering all of the vegetation of eight feet and lower. The uniformity of application was solely dependent upon the judgment of the two-man crew because access to the sites was very restrictive.

The methods of application used in this project were designed to simulate land application of liquid sewage sludge by conventional spray irrigation. Therefore, the evaluation of

equipment utilized in this project was never conducted. The anticipated evaluation of equipment used for direct incorporation of sewage sludge was also not accomplished because of changes in cropping programs and the unavailability of higher solids content sludge.

LOADING AND DISTRIBUTION OF WASTEWATER SOLIDS

Dried cake and liquid sludge were applied at the two Webbs Mill sites (Lakewood and Woodmansie) and at the Colliers Mills (Downer) location on all cleared 44.8 and 89.6 metric ton/hectare/year experimental plots. However, only liquid sludge was applied to the 22.4 metric ton and natural vegetation plots at the three application sites. The cake was spread on May 29-31, 1973 at the Webbs Mill sites and on June 11, 1973 at the Colliers Mills plots. The cake was used as a supplement to liquid sludge to insure a nutrient supply before the initial planting of crops. No additional applications of cake were spread following its initial use in the Spring of 1973.

As discussed in a previous section, it was observed that the cake had accumulated a considerable amount of sand from the sludge drying beds and from transfer piles at the application site. Since the annual loadings were based on dried tons of wastewater solids, it became necessary to estimate the dried weight of cake, excluding sand, in order to equate the cake and liquid sludge loadings to the cumulative solids loading. This value was determined by relating the total volatile solids content of the cake drying on the Lakewood sludge drying beds to that of the cake applied to the application plots. Assuming that the percent volatile solids content is constant for both cakes, one can estimate the quantity of sand accumulated in the field cake. In general, the cake did not supply as much nitrogen, phosphorus and potassium as anticipated as compared to the liquid sludge, due to the unexpected accumulation of sand.

Liquid sludge was transported to the application site in a 3000 gallon tank truck. The wastewater solids were then transferred to a 1500 gallon field unit for application to individual experimental plots.

A record of the total wastewater solids and individual sludge constituents applied to each experimental plot was compiled to evaluate environmental impact of sludge constituents with regard to the soil, vegetation and groundwater. Cumulative pounds of sludge components (dry weight basis) for the 1973, 1974, and 1975 applications are tabulated. Based on the designated annual tonnage (22.4, 44.8, and 89.6 t/ha/y) of wastewater solids applied to respective experimental plots, most plots were within a few percent of the design loadings.

The data were tabulated and transformed to express the metric loadings of wastewater solids and individual sludge constituents for three years of solids application.

The total loadings of a few sludge constituents applied to individual plots are very low, particularly mercury, cadmium, chromium and nickel applied after three (3) years are 0.6, 5.1, 15.4, and 11.5 kg/ha respectively. From a simple quantitative viewpoint, it would appear that the low loadings of these sludge constituents would cause no significant change or "measurable" impact to the soil, vegetation or groundwater. The loadings of all other constituents are at least an order of magnitude greater than the above heavy metals. Therefore, a "measurable change" or environmental impact can be detected and evaluated in the soil-vegetation-groundwater system for all sludge constituents except the trace elements - mercury, cadmium, chromium, and nickel.

SECTION 7

AGRONOMIC ASPECTS

GROWTH OF NATIVE VEGETATION

Sewage sludge was sprayed on the native vegetation from a manifold of three nozzles mounted on the field-spraying trailer at a height of 8 feet. This resulted in a rather complete coverage of all foliage beneath that height. This coverage appeared to be sufficient to effectively shut off photosynthesis and cause an appreciable elevation in leaf temperature because of the albedo effect. Indeed, it was not uncommon to see numerous leaves (primarily oak) which had been sludge covered that became necrotic on the stem in mid-summer. Sludge applications were made to native vegetation plots three to six times per year for the three year period. At times months elapsed between applications. Rainfall usually removed the sludge from the foliage readily. The duration of restricted photosynthesis would be extremely variable and not necessarily a discrete phenomenon. The foliage and stems of the low-growing huckleberry bushes were frequently completely covered by sludge, however, loss of such plants appeared to be at a very low incidence.

Data reflecting the growth of native woody species taken in late 1974, after one and one-half seasons of sludge application, indicate that the nutrient value of the sludge had a positive effect that exceeded any detrimental effect (due to toxicity from heavy metals and/or inhibition of photosynthesis) on native vegetation.

Plot locations were selected at two sites in Ocean County to include three soil types typical of the coastal plain and Pine Barrens. At each location, 0.94 ha plots of native vegetation were designated to receive sludge applications. The 0.94 ha plots received 44.8 t/ha/y dry matter in 1973 and 1974, and responses were measured of the natural vegetation to the sludge applications during and after the second year of sludge induced growth, the summer of 1974 and the following winter dormant period.

Growth of Woody Species

Increment borer cores were taken from randomly selected trees in sludge-treated plots and also in adjacent check areas. Annual growth ring width of oak species showed an increase due to sludge applications of the previous year on the Downer loamy sand. The growth of scarlet oak in the control series was less in 1974 than in 1973, but in the sludge-treated area ring width growth increased in 1974. With only ten samples taken from each test plot no statistically significant difference between means of increment change could be proved by a "F" test of unpaired samples.

When sections of twigs were taken from 25 randomly selected oaks (with trunk diameters of approximately 5 cm) from the sludge-treated plot and also the surrounding native vegetation on Lakewood sand, the increase in mean ring growth due to sludge treatment was significant at the 5 percent level of probability.

When oaks were similarly sampled from Woodmansie sand a sludge induced increase in growth ring width was significant at the 1 percent level of probability.

Cross-sections of main stems of huckleberry bushes, 25 from each test plot, on each of the three soils were similarly measured. The positive response in ring width increase due to sludge treatment was significant at the 5 percent level of probability for plants on the Downer and Woodmansie site, and at the 1 percent level for plants on Lakewood sand. The foliage of sludge-treated huckleberry bushes was completely covered with sludge after each spraying. The positive response of these small plants to sludge treatment is somewhat more surprising than is the response of the taller oaks, some of whose foliage would escape the temporary photosynthesis-blocking treatment.

Insufficient numbers of pines were found on the site of the Downer soil. On both the Lakewood and Woodmansie sand sites increment borings were taken from 25 pitch pines in sludge-treated and untreated areas. On both sites the increases in growth ring width of pines due to sludge treatment were significant at the 1 percent level of probability.

Some dead plants of all species were found within sludge-treated and untreated plots at all three soil sites. The incidence of injury did not appear to be different because of sludge application. The re-sprouting from the base of oaks in treated plots was readily consumed by wildlife.

Results of Tissue Analyses of Native Species

The color of the foliage of native vegetation receiving 44.8 t/ha/y of sewage sludge was darker green than in the same species surrounding these plots. This was evident on all three soil types in 1974 through 1976. The color change was typical of a response to nitrogen fertilization. Data indicates a consistent increase in the content of total Kjeldahl nitrogen for the four native species sampled.

The foliage sample size did not prove adequate for total Kjeldahl analyses on these species for all soils and all years. Therefore statistical analyses of the above data was not possible. Analyses of P, K, Ca, Mg, Zn, Cu, Mn, Ni, Pb, and Cr were obtained for oak, huckleberry, and pine under all these conditions, and the data were subjected to analyses of variance. Table 7 indicates that the foliage of all three plant species growing in sludge treated plots contained significantly more P than the no-sludge counterparts surrounding these plots. Only the sludge-treated oak leaves contained significantly more K than the no sludge checks. Calcium concentration was lower in foliage of all three species, but significantly so only in huckleberry.

No significant differences (at the 5% level of probability) were observed in the content of Mg and the trace elements and heavy metals in the foliage of these species. Mean values for Zn, Cu, and Pb were higher for the sludge treatment of all three species. The Mn and Cr contents were appreciably lower in oak and huckleberry on plots that received sludge.

These soils were not limed. If they had been, the Ca and Mg contents of the tissue might have been higher and the heavy metals might have been lower because of the latter's lower solubility at higher soil pH. The observation previously cited, viz, increased growth of native vegetation following sludge application, is consistent with the increased content of N, P, and K in the foliage.

EFFECTS OF WASTEWATER SOLIDS APPLICATION ON CROPS

Plant Establishment

Midland bermudagrass, newly planted on experimental sites, grew through applications of wastewater solids sprayed on the surface at intervals varying from daily for three consecutive days to weeks without an application. A heavy initial application, such as the 89.6 t/ha/y Downer plot received one month after being planted in 1973, was equivalent to 288,960 l/ha of sludge in three days. At a mean of 5.68% solids, 16.38 t/ha of dry matter equivalent was applied. This was the equivalent to a depth of liquid sludge averaging 96 mm/day for three days.

TABLE 7. CONCENTRATION OF ELEMENTS IN FOLIAGE OF NATIVE SPECIES WITH AND WITHOUT SLUDGE. MEANS OF THREE SOILS FOR THREE YEARS

	P	K	Ca %	OAK				Cu	Mn	Ni	Pb	Cr
				Mg	Zn	ppm						
no sludge	.101	.63	.49	.111	35.6	19.7	1198	2.8	4.8	2.51		
sludge, 44.8 t/ha/y	.157	.70	.42	.190	43.1	20.5	659	4.7	5.5	1.72		
statistical significance	*	*	NS	NS	NS	NS	NS	NS	NS	NS		
HUCKLEBERRY												
no sludge	.076	.55	.80	.166	18.1	12.0	357.6	3.8	5.4	2.69		
sludge, 44.8 t/ha/y	.228	.63	.53	.194	40.1	25.3	139.5	3.3	6.9	1.42		
statistical significance	*	NS	*	NS	NS	NS	NS	NS	NS	NS		
PINE												
no sludge	.101	.54	.31	.093	46.9	12.0	177	5.1	4.3	2.36		
sludge, 44.8 t/ha/y	.196	.80	.24	.093	82.4	26.9	180	6.9	8.9	2.91		
statistical significance	*	NS	NS	NS	NS	NS	NS	NS	NS	NS		

*Indicates means are different at the 5% level of probability. NS = not significant at 5%

While occasional leaves of bermudagrass were submerged by the sludge, the technique of plantings on ridges between tractor tire tracks prevented serious inundation of plants. Essentially no leaf scorch or injury of any sort to the bermudagrass or other volunteer plant species could be attributed to sludge applications.

Tomatoes grew profusely after following initial applications of sludge. They grew well, flowered, bore fruit, and showed negligible evidence of injury from subsequent applications of sludge sprayed over them.

Fall panicum (Panicum dichotomiflorum), a summer annual grassy weed volunteered profusely on the 44.8 and 89.6 t/ha/y plots that had received a portion of the first year's sludge as filter-bed dried cake plowed-in prior to planting the Midland bermudagrass. Apparently, this weed proliferated the previous season on the sand-bed filter and dispersed large quantities of seed over this sludge. Essentially none of this weed appeared in any of the 22.4 t/ha/y plots nor in the native vegetation plots both of which received 44.8 t/ha/y of liquid sludge only.

In addition to the abundance of tomato and fall panicum seedlings, there were watermelon, muskmelon, cucumbers, peanuts, and citrus growing in the sludge treated plots. Common weeds in these plots included common ragweed (Ambrosia artemisiifolia), lambsquarter (Chenopodium album), redroot pigweed (Amaranthus retroflexus), purslane (Portulaca oleracea), smartweed (Polygonum pennsylvanicum), crabgrass (Digitaria sanguinalis), black nightshade (Solanum nigrum), goosegrass (Eleusine indica), and carpetweed (Mollugo verticillata). These plants were not found in surrounding areas that had not received sludge, for these areas had only recently been cleared of pine barrens forest that did not include these species.

The site of the Downer soil (at Colliers Mills) was near a long-established paved road, and a pine barrens sedge (Carex pennsylvanica), and pokeweed (Phytolacca americana) volunteered in both the treated plots and in the cleared margins.

The only plant that successfully competed with the Midland bermudagrass following its establishment on sludge-treated plots was fall panicum. Properly timed harvesting could favor the development of Midland bermudagrass over the fall panicum, and this weed would not jeopardize production of the crop species.

Each fall, soon after the last harvest of bermudagrass for the season, the plots were sown to common rye (Secale cereale). A grain drill was modified so that opening coulters cut through the dormant sod of bermudagrass and rye seed was placed approximately 4 cm deep in rows 22 cm apart. At a

seeding rate of 100 kg/ha rye emerged uniformly each year even when drilled into freshly applied sludge or when sludge was subsequently applied over the seeding. A substantial crop of rye would develop in early spring. Midland bermudagrass would re-establish dominance after rye was harvested.

On July 17, 1976, 'Piper' sudangrass was sown at 50 kg/ha with the same drill described above. Low rainfall following seeding resulted in marginal stands of sudangrass, especially where tillage was not employed.

Crop Production

Yield determination from the 0.093 hectare plots were made by harvesting the grass with a commercial forage harvester having a 1.8 m cutter bar. A full swath was taken from at least one length of the plot (61 m); when yields were low two lengths were harvested. Typically, about 50 kg of the chopped grass was blown into the box body of a following truck where it was collected and weighed. A subsample of approximately 500 gm was taken from five locations in the pile and sealed in a plastic bag until the wet weight of the sample was determined. After attaining a constant weight in a 70°C forced air dryer, percent dry matter was calculated in order to determine plot yield. A portion of this sample was ground through a 40 mesh stainless steel screen for chemical analyses.

Yields from the no-sludge control plots (9m x 3m) were determined by harvesting 8 m lengths with one of several sickle-bar-mowers that cut a width slightly less than 1 m in width calculating the exact area harvested, and applying data from its samples, handled as above, to determine dry matter and chemical analyses.

Marked and consistent increases in yield of dry matter of Midland bermudagrass occurred with increasing rates of sludge applied during 1973, the year of establishment. It was noted that green color of the grass was darker at the higher rates of sludge application. The low rate actually appeared N deficient. Yields are low, however, because (1) planting was two months late (due to the need to get stable background data from well water samples), (2) height of cut had to be exceptionally high to avoid rough-terrain damage to the field chopper used in harvesting, and (3) less than three quarters of the first year's rate of sludge had been applied at the time of the first harvest.

The superior initial stand of Midland bermudagrass on the Downer soil enabled an earlier harvest, and a second harvest the first year. Hence, substantially higher total yields were obtained on the Downer soil than the Lakewood and Woodmansie soils. Wildlife consumption of grass was negligible.

The yields of rye harvested from the Downer plots in the spring of 1974 were not proportionate to the intended rates of application as shown in Table 8. It can be calculated that ratios of rates of application were approximately 1:1:2 rather than the target ratios of 1:2:4. The fact that the high rate plot received a depth of 4.62 cm sludge between March 13, 1974 and March 20, 1974 when the rye was just breaking dormancy may account for its relatively low yield. The stand of rye in the high rate plot appeared sparser but darker green than those in other plots at the time of harvest.

Rye on Lakewood and Woodmansie soils was consumed completely by wildlife, and no estimates of yield were possible.

Yields of bermudagrass similarly did not relate directly to stated rates of application. Sludge application records indicate that rates applied did approximate target values. For the first harvests of Midland bermudagrass in 1974 the timing of applications was good in that they occurred largely after the grass broke dormancy and well before harvest time. Yields of all plots were good, but did not increase in proportion to rate of sludge applied.

In the second harvest of Midland bermudagrass the inverse relationship of yield to rate of sludge application seen in Table 8 for the Lakewood soil is attributed to the fact that too much sludge was applied too late. Lodging was excessive in the high rate plots, yields were reduced, and the grass was injured.

Yields of third harvests were limited by low temperatures. It was imperative that grass be harvested in the fall to facilitate drilling in the rye cover crop.

In 1975 rye was again harvested from the Downer soil plots, as indicated in Table 9 but consumption by wildlife was total on the Lakewood and Woodmansie soils. Rainfall was exceptionally high, and yields of bermudagrass from plots receiving the lowest rate of sludge (22.4 t/ha/y) were down slightly lower than those of the previous year. With the exception of the Woodmansie soil, yields of plots of the highest sludge rate (89.6 t/ha/y) were higher. The records of sludge applications and further calculations indicate that 40% of the year's allocation was made after the last harvest. But the highest yields obtained from these sludge plots, in a year with good rainfall, was still far below expectations, being approximately half that obtained from research plots receiving commercial fertilizer.

No sludge was applied in 1976; the intent being to evaluate the residual effects of the applications of the three previous years. Table 10 indicates that rye yields were appreciably

TABLE 8. GRASS HARVESTED FROM SLUDGE PLOTS IN 1974

Soil	Crop	Date of harvest	t/ha/y of sludge, dry matter equivalent	kg/ha dry matter harvested
			22.4	44.8
				89.6
<u>DOWNER</u>				
	rye	5/7	1,179	1,683
	bermudagrass	7/11	4,161	4,019
	bermudagrass	9/6	1,915	2,963
	bermudagrass	10/17	582	938
Total harvested			7,837	9,603
<u>LAKEWOOD</u>				
	rye	7/12	3,771	3,464
	bermudagrass	9/5	3,536	2,942
	bermudagrass	10/18	826	639
Total harvested			8,133	7,045
<u>WOODMANSIE</u>				
	rye	7/12	3,811	4,437
	bermudagrass	9/5	3,332	2,063
	bermudagrass	10/18	808	631
Total harvested			7,951	7,131
				8,271

TABLE 9. GRASS HARVESTED FROM SLUDGE PLOTS IN 1975

Soil	Crop	Date of harvest	t/ha/y of sludge, dry matter equivalent	
			22.4	44.8
kg/ha dry matter harvested				
DOWNER				
	rye	5/21	1,233	2,112
	bermudagrass	7/10	1,732	1,349
	bermudagrass	8/14	1,442	2,380
	bermudagrass	10/16	2,500	1,575
				1,363
				3,418
				2,596
				3,113
Total harvested			6,907	7,416
				10,490
LAKEWOOD				
	rye			consumed by wildlife
	bermudagrass	7/1	2,748	3,660
	bermudagrass	7/29	1,771	1,259
	bermudagrass	9/15	1,762	2,539
				3,282
				2,255
				2,511
Total harvested			6,281	7,458
				8,048
WOODMANSIE				
	rye			consumed by wildlife
	bermudagrass	7/1	3,473	3,716
	bermudagrass	7/29	1,893	1,319
	bermudagrass	9/15	1,058	2,203
				1,054
				2,841
				2,921
Total harvested			6,424	7,238
				6,816

TABLE 10. GRASS HARVESTED FROM SLUDGE PLOTS IN 1976

Soil	Crop	Date of harvest	t/ha/y of sludge, dry matter equivalent	kg/ha dry matter harvested
			22.4	44.8
				89.6
<u>DOWNER</u>				
	rye	6/1	602	893
	sudangrass	9/29	2,322	979
				1,916
	Total harvested		2,924	1,872
				2,803
<u>LAKWOOD</u>				
	rye	-----	-----	consumed by wildlife -----
	sudangrass	9/29	1,085	1,065
				1,753
<u>WOODMANSIE</u>				
	rye	-----	-----	consumed by wildlife -----
	sudangrass	9/29	1,073	1,688
				4,160

lower than in previous years when some sludge was applied in the spring. No increase in yield due to the highest sludge rate can be seen, but the intensity of green color of the rye was improved with increasing sludge rates. Deer grazing appeared to increase with sludge application rate. This factor was not discernable the first year, unimportant the second year, but appreciable the third year. Deer had found a source of food they like and frequented the area more with time, eating more of the rye in the heavily treated plot. The greater wildlife population on the sites of the Lakewood and Woodmansie soils again completely consumed all the rye from all of those plots.

Unlike previous years the Midland bermudagrass did not recover. Winter killing was quite complete on the Lakewood and Woodmansie soils, and very extensive on the Downer soil. Piper sudangrass was drilled in as previously described and yields were taken to measure residual effects of the sludge on all sites. Again yields did not correlate well with sludge rate applied. Some deer grazing of sudangrass could be detected in plots on the Downer site, but grazing was so intense on the Lakewood and Woodmansie soils as to require estimates of consumption to adjust yields.

After the sudangrass was established, and much later than Midland bermudagrass normally appeared, the few surviving plants of bermudagrass began to show a pattern of survival. Many more plants survived on the Downer loamy sand than on the Lakewood sand or the Woodmansie sand. More bermudagrass survived in plots receiving lower sludge rates or were outside of the plots receiving sludge than in those plots, and more bermudagrass survived around obstacles where bermudagrass could not be mowed readily and therefore was defoliated less frequently.

It would appear therefore that survival of bermudagrass was enhanced by a narrower N:K ratio and a better carbohydrate status.

CHEMICAL CONTENTS OF PLANT TISSUES

Subsamples of Midland bermudagrass taken from the large groundwater monitored plots were analyzed for major, secondary, and minor nutrient elements, as well as heavy metals. Sludge contamination of foliage was apparent on certain samples, and concentration of chemical elements proved to be elevated in those instances. Where no sludge had been applied since last harvest, exceptionally low values were obtained. Such conditions resulted in major variations of these elements in samples analyzed.

Variations in dry matter yield of harvested forage were often major factors in determining quantities of elements removed for individual harvests on each of the three soils over

the study period.

Although elemental composition of crops relate well to rate of sludge applied, exceptions among certain heavy metals are occasionally found. The amounts of nutrients removed by cropping are small in comparison with amounts applied in sludge and even with those remaining on the soil surface in the form of dried sludge. Availability of residual nutrients to the last crop, sudangrass, proved to be very low.

RESPONSE OF A COOL SEASON GRASS MIXTURE TO SLUDGE APPLICATIONS

Sludge was applied by spray-boom application (previously described) to a mature stand of a mixture of cool-season perennial grasses. The grasses had been established with conventional techniques including liming and fertilizing by the New Jersey Division of Fish, Game, and Shellfisheries on Lakewood sand. In a randomized block design three replicates of 7.5 m x 13.5 m plots were established to receive 0, 44.8, and 89.6 t/ha/y dry sludge matter equivalent usually from mid-March to the end of October, 1973, to 1975 inclusive. One replicate was fenced to prevent grazing by deer. No harvests were recorded in 1973 but three were taken in 1974 and 1975, and one in 1976 to evaluate residual effects. Yields were determined by harvesting strips approximately 1 m wide through the center of each plot and correcting for moisture content. Subsamples were analyzed for elemental composition as previously described. In November 1975, 200 kg K/ha from KCl was applied to 2 m x 6 m portions of each plot.

Yield Responses

Evidence of grazing of these grasses by wildlife was particularly noticeable in the cool seasons as was reported for other situations by Kalmbacher. Only during the warmer months was grazing pressure reduced to the point where forage would accumulate to permit harvesting of unfenced plots. Grazing was most intense on plots receiving the highest rate of sludge, and grazing increased annually as if more wildlife was utilizing the plots with each succeeding year. In the last year (1976) the density of rabbit pellets was estimated at 100/m² on a closely grazed plot that had received 89.6 t/ha/y for 3 years. Rabbit pellets far exceeded deer pellets. Rabbits, but not deer, did enter the fenced area at times but the fenced area was generally grazed less than unfenced plots.

Differential application of sludge, and hence grazing also brought about changes in population of grass species in these plots. Table 11 indicates that species not adapted to high fertility and close grazing were being depleted from plots receiving higher rates of sludge. Kentucky bluegrass continued

TABLE 11. PLANT SPECIES COMPOSITION AS AFFECTED BY GRAZING AND SLUDGE APPLICATION
ON LAKEWOOD SAND IN 1974

	Unfenced		Fenced	
	Species	% Composition	Species	% Composition
<u>No Sludge</u>				
	Andropogon spp.	80	Andropogon spp.	40
	Tall fescue	10	Tall fescue	30
	Orchardgrass	5	Kentucky bluegrass	5
	Fine fescues	2	Fine fescues	3
	Kentucky bluegrass	2	Weeds	2
	Weeds	1	Orchardgrass	20
<u>44 t/ha/y</u>				
	Tall fescues	77	Orchardgrass	50
	Kentucky bluegrass	20	Kentucky bluegrass	30
	Orchardgrass	3	Tall fescue	20
<u>89.6 t/ha/y</u>				
	Kentucky bluegrass	60	Orchardgrass	85
	Tall fescues	40	Kentucky bluegrass	10
			Tall fescue	5

to increase and the tall fescue and orchardgrass decrease, under high fertility and close grazing to the extent that in 1976 the unfenced 89.6 t/ha/y plots were essentially devoid of other plants. During cool seasons in particular, and with ample moisture available, these plots resembled a good lawn.

A positive response to sludge application rate is seen in terms of 1974 yield data in Table 12 particularly for the fenced plots. Yields are not as high as they should be because there was some grazing by rabbits even in fenced plots, and timing of application of the sludge was not the best. These same trends are seen in yields of 1975 shown in Table 13. Moisture was more abundant in 1975 and should have induced better growth, but grazing by wildlife was increasing, and it was suspected that potassium deficiency might also have been limiting production.

Another harvest was made on May 28, 1976 to measure the residual effects of sludge applications that were applied prior to the previous fall, and to determine the response to supplemental potassium fertilization of the previous fall. Yields increase only slightly because of previously applied sludge except where potassium was supplemented at 200 kg/ha. Consistent increases in yield due to potassium applications resulted. The increase due to potassium is uniform where no sludge was applied because grazing was minimal. Increases of 153 and 128% were noted where potassium treated plots were fenced. Similar increases due to potassium were recorded in unfenced plots where, by chance, grazing was not so intense.

The positive response of these cool season grasses to surface applied sludge was impressive, but could have been greater if potassium were supplemented earlier. Better timing of application of sludge may have enhanced yields also. However, one application of approximately 6 cm in mid-summer merely retarded growth, but did not cause direct injury to these grasses. Limiting the severity of defoliation might have assured persistence of the taller, usually more productive, tall fescue and orchardgrass.

Chemical Analyses of Cool Season Grasses

A single set of samples was taken for analyses from all plots in 1973, while the first two harvests and all three harvests were sampled and analyzed in 1974 and 1975, respectively. Control plots without sludge are particularly low in nitrogen content and crude protein, but as can be seen in Table 14 the mean values of three replicates of samples from plots receiving 44.8 t/ha/y are appreciably increased for every harvest of the three year period. Increasing the rate of sludge

TABLE 12. YIELDS OF COOL-SEASON GRASS MIXTURE PLOTS ON LAKEWOOD SAND, 1974

Plot No.	Dry sludge Kg/ha/y	Condition	Kg/ha dry matter harvested			Total
			June 6	Aug. 20	Oct. 6	
2	0	grazed	742	379	136	1,257
6	0	grazed	1,110	985	146	2,241
7	0	fenced	1,643	1,527	759	3,929
8	44.8	grazed	1,770	660	2,539	4,969
4	44.8	grazed	1,850	613	3,076	5,539
3	44.8	fenced	4,103	3,086	2,074	7,189
1	89.6	grazed	641	11	243	895
9	89.6	grazed	862	11	1,808	2,681
5	89.6	fenced	3,981	3,428	2,075	9,484

TABLE 13. YIELDS OF COOL-SEASON GRASS MIXTURE PLOTS ON LAKEWOOD SAND, 1975

Plot No.	Dry sludge t/ha/y	Condition	Kg/ha dry matter harvested			Total
			June 18	Aug. 13	Oct. 28	
2	0	grazed	482	471	867	1,820
6	0	grazed	80	623	94	797
7	0	fenced	94	1,316	787	2,197
8	44.8	grazed	1,340	1,757	450	3,547
4	44.8	grazed	3,110	2,295	847	6,252
3	44.8	fenced	1,901	2,342	94	4,337
1	89.6	grazed	1,690	1,363	215	3,268
9	89.6	grazed	2,768	1,718	327	4,813
5	89.6	fenced	2,206	2,628	458	5,292

TABLE 14. CHEMICAL ANALYSES OF COOL-SEASON GRASSES OVER A 3-YEAR PERIOD AS AFFECTED BY SLUDGE RATES

Sludge Rate t/ha/y	Percentage (%)					K	Ca	Mg
	N	Protein	P	S	Cl	Na		
0	1.37	8.6	0.75	1973 Harvest				
44.8	2.73	17.1	0.98	0.16	0.40	0.03	1.33	0.10
89.6	4.09	25.6	1.00	0.35	1.04	0.06	1.70	0.17
				0.35	1.67	0.17	2.75	0.26
0	1.12	7.0	0.45	1974 - First Harvest				
44.8	2.49	15.5	0.62	0.11	0.32	0.02	1.08	0.06
89.6	2.77	17.3	0.50	0.17	1.09	0.06	1.88	0.15
				0.19	1.10	0.08	1.88	0.16
0	1.31	8.2	0.36	- Second Harvest				
44.8	2.35	14.7	0.70	0.10	0.29	0.02	1.56	0.09
89.6	2.49	15.5	0.50	0.21	0.60	0.09	1.88	0.20
				0.23	0.64	0.09	1.96	0.20
0	1.21	7.6	0.48	1975 - First Harvest				
44.8	2.76	17.3	0.62	0.13	0.35	0.03	1.05	0.16
89.6	2.96	18.5	0.65	0.21	0.74	0.16	0.98	0.26
				0.22	0.79	0.18	1.20	0.27
0	1.17	7.3	0.40	- Second Harvest				
44.8	3.45	21.6	0.77	0.14	0.33	0.03	0.92	0.18
89.6	3.44	21.5	0.77	0.23	0.48	0.08	0.82	0.28
				0.24	0.58	0.11	1.08	0.29
0	1.55	9.7	0.48	- Third Harvest				
44.8	3.44	21.6	0.67	0.17	0.39	0.03	0.94	0.18
89.6	3.80	23.8	0.73	0.27	0.64	0.09	0.95	0.28
				0.29	0.67	0.12	1.12	0.29

TABLE 14. (cont'd.)

Sludge Rate t/ha/y	Fe	Mn	Cu	Parts per million (ppm)				Cr	Cd	Pb
				Zn	Ni					
0	2330	20	39	1973 Harvest				6	3	3
44.8	7000	60	156	95	4			30	5	3
89.6	7000	83	156	383	7			23	4	3
0	150	11	31	1974 - First Harvest				3	3	3
44.8	170	18	38	19	4			4	4	3
89.6	570	37	42	46	4			6	4	3
				100	4					
0	40	15	11	- Second Harvest				3	3	3
44.8	220	50	19	25	4			8	3	3
89.6	340	57	21	68	4			8	3	3
				78	4					
0	230	11	15	1975 - First Harvest				8	3	2
44.8	1135	36	37	42	5			11	3	2
89.6	1323	53	37	195	7			13	3	2
				205	12					
0	80	13	7	- Second Harvest				12	3	2
44.8	426	56	37	38	5			15	3	3
89.6	1136	86	39	159	7			15	3	3
				210	15					
0	330	20	17	- Third Harvest				8	3	2
44.8	1468	77	34	48	4			10	3	3
89.6	2250	110	37	203	12			10	3	3
				395	18					

to 89.6 t/ha/y increased the nitrogen content slightly above the 44.8 t/ha/y rate. A similar pattern of a greatly increased content with the first increment and a moderate increase with the second increment include Na, K, Fe, Mn, Zn and Ni (the latter especially so in the last year). Essentially no increase was seen in Cd and Pb contents of this grass tissue as a result of rates of sludge applied.

The increase in heavy metal content of crops appears to be within allowable limits as to phytotoxicity and mammalian toxicity. The increases in nutrient content of this forage due to sludge applications confirms the fertilizer value of the sludge and should prove to be of appreciable practical significance in the production of wildlife or livestock.

SOIL WATER

During 1974, while the second year's application of sewage sludge was being surface-applied, measurements of soil water were made on four occasions. Neutron probe access tubes were installed according to standard procedures in the Lakewood sand and water content was measured with a Troxler neutron scaler and probe. Areas considered were those 1) planted to Midland bermudagrass and treated with each of the three rates of sludge, 2) planted to bermudagrass but receiving no sludge, and 3) bare soil, no vegetation and no sludge. The latter two types of plots were always located in close proximity to each of the sludge-treated plots.

The data of July 17, 1974, in Figure 22 were taken three days after a harvest and five days after 15 mm of rainfall. At each level of sludge applied, moisture had been markedly diminished from the entire soil profile by the sludge-stimulated bermudagrass. Where the soil surface was kept bare, appreciably more moisture was detected at the intermediate and greater depths. The plots of bermudagrass that received no sludge, and only a minimum of fertilization to assure survival, had an intermediate level of moisture left in the soil profile. Obviously, the production of a vigorous crop of bermudagrass required an appreciable withdrawal of moisture from the soil profile.

Data taken 18 days after harvest and six days after a rainfall of 71.5 mm showed somewhat less discrete moisture curves due to type of cover and sludge rate. More moisture was retained in the 0-305 mm depth zone where the heavy rate of sludge was applied. A less marked effect of this sort is shown in Figure 23 for the intermediate sludge rate and is essentially non-existent for the lowest sludge rate. At this

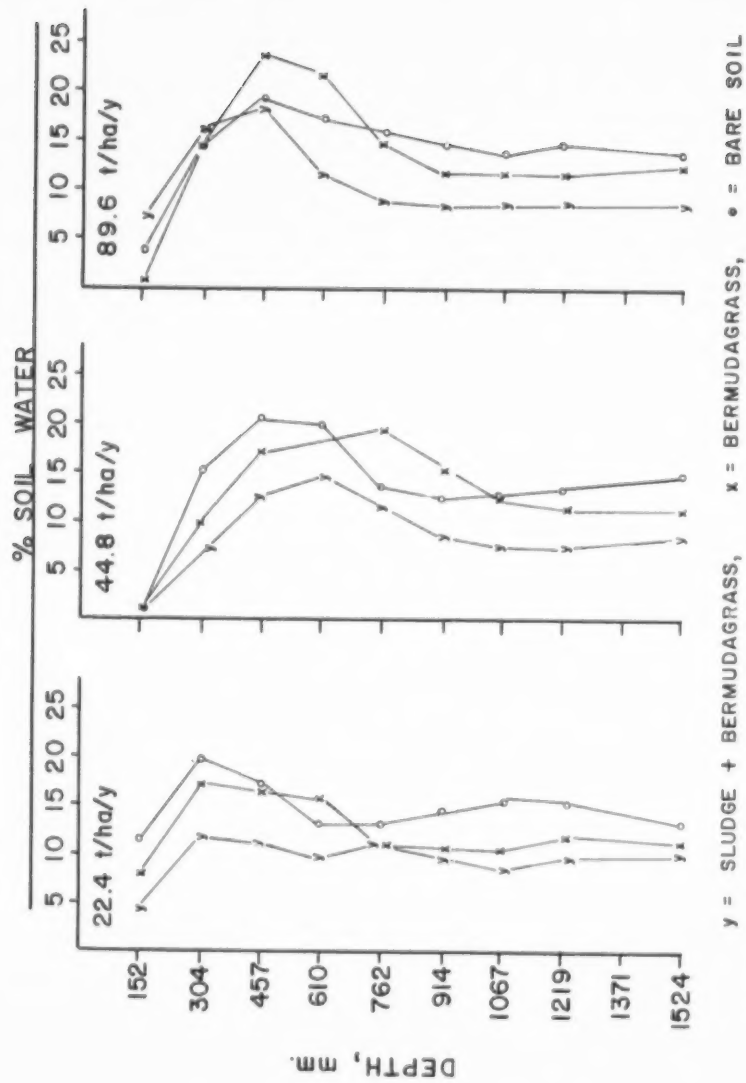


Figure 22. Soil water variations with depth, determined by neutron probe under various conditions of Lakewood Sand on July 17, 1974.

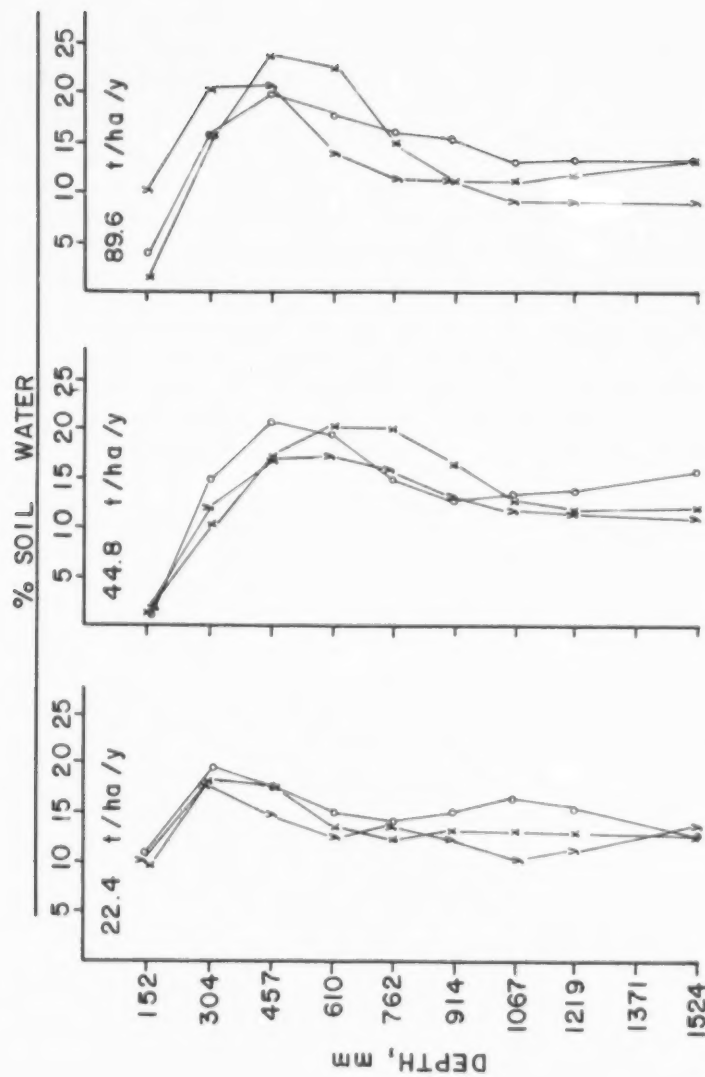


Figure 23. Soil water variations with depth, determined by neutron probe under various conditions of Lakewood Sand on July 31, 1974.

time crop depletion of moisture was not paramount, but the surface mulch effect of the sludge was a factor.

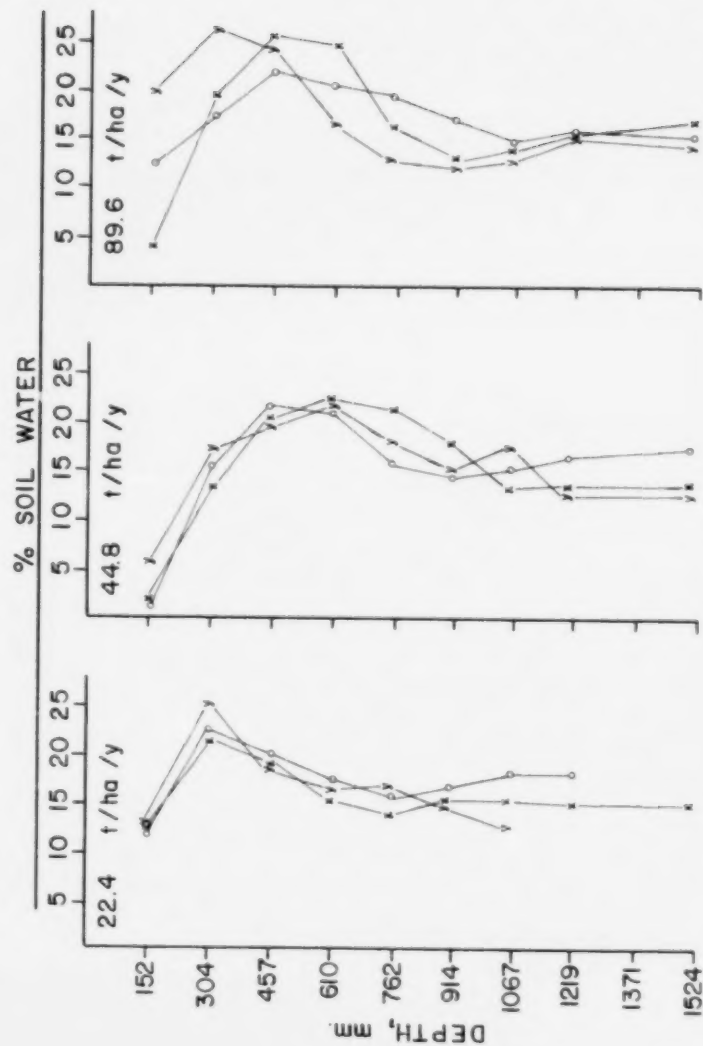
Soil moisture measurements taken five days after a second harvest of Midland bermudagrass and four days after a 31.3 mm rain are shown in Figure 24. Increasing the rate of sludge applied to 80.6 t/ha/y resulted in more moisture retained in the 0-304 mm depth of the profile than was evident at lower rates or other covers. The lower moisture values for the above treatment that are evident at the 610-762 mm depth may be a result of the moisture depletion by the sludge-stimulated crop that was not restored by the previous rain.

The data in Figure 25 were taken 16 days after a rainfall of 15 mm, and 36 days after the previous harvest. Mid-October temperatures were limiting bermudagrass production and a modest harvest was taken four days later. The bermudagrass from the intermediate sludge rate depleted moisture from the 304 to 610 mm depth. Moisture was high in the 0 to 304 mm depth zone under the high sludge rate possibly due to the accumulation of a wet mulch of sludge on this surface. In this same plot the bermudagrass appeared to extract appreciable moisture resulting in decidedly dryer soil in the 610 to 1219 mm depth.

On July 17, 1974, sludge applied at 44.8 t/ha to native vegetation on Lakewood soil increased soil moisture to a depth of just over 1 m, but beneath this point soil moisture was somewhat lower than a comparable non-treated area. Later, on July 31, 1974, the sludge-treated forest soil had higher soil moisture contents to a depth of just under 1 m, and beyond this depth soil moisture decreased. Similar trends with identical breaking points, at a depth between 914 mm to 1067 mm, were recorded on September 10, 1974 and October 10, 1974. (See Table 16).

SOIL TEMPERATURE

Data in Table 15 show that temperatures of forested Lakewood soil were appreciably lower than bare soil, and bermudagrass covered soil (with or without treatment) from the surface to a depth of 305 mm on July 10, 1974. Bare soil was warmer through this profile than were plots of soil with a bermudagrass cover. Increasing rates of surface applied sludge to bermudagrass for 2 years tended to decrease soil temperature especially through the first four increments of profile depth at this date. Similarly, on July 31, 1974 the surface application of sludge to Lakewood soil covered with bermudagrass decreased soil temperatures to a depth of 305 mm. Higher rates of sludge had no greater effects on soil temperatures than the lowest rate. The same was true on the last date of measurement. The trend toward decreasing temperature with increasing depth was greatest at the earliest date and least at the latest date.



v = SLUDGE + BERMUDAGRASS, x = BERMUDAGRASS, o = BARE SOIL
 Figure 24. Soil water variations with depth, determined by neutron probe under various conditions of Lakewood Sand on September 10, 1974.

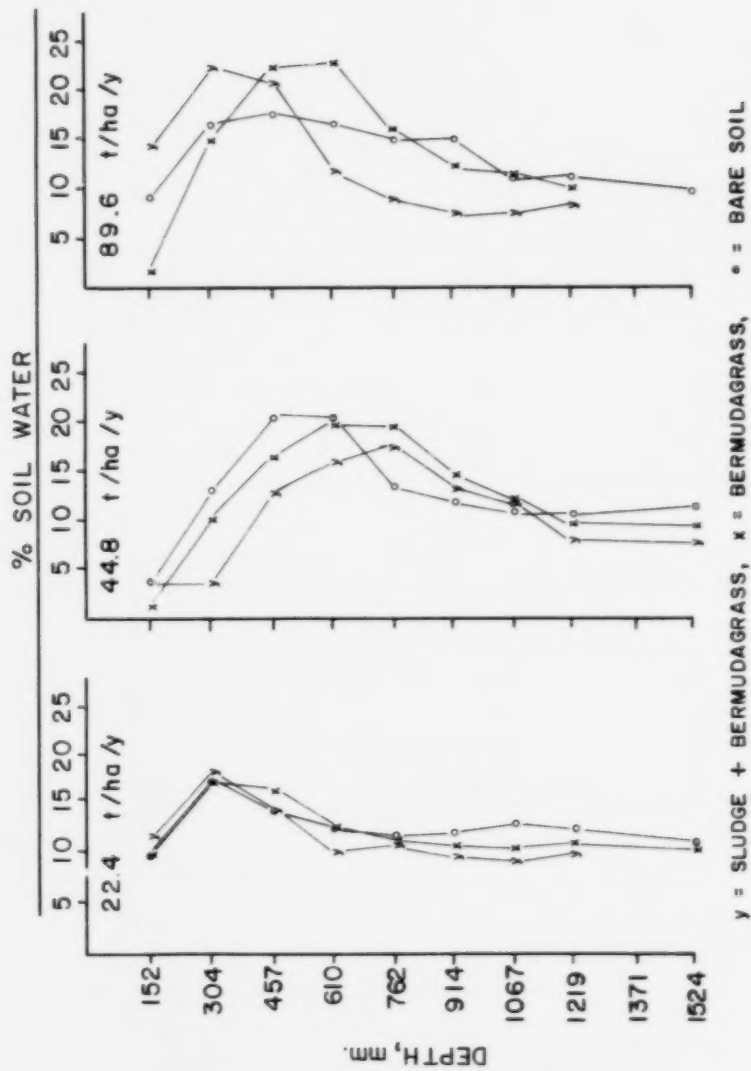


Figure 25. Soil water variations with depth, determined by neutron probe under various conditions of Lakewood Sand on October 10, 1974.

TABLE 15. TEMPERATURE OF LAKEWOOD SAND AT VARIOUS DEPTHS AS AFFECTED BY COVER AND FIRST YEAR'S APPLICATION OF SLUDGE

Cover	t/ha Sludge	July 10, 1974							
		0	51	102	mm depth				305
Forest	0	23*	23	20	20	20	19	19	19
Bare soil	0	38	34	38	34	29	26	24	24
Bermudagrass	0	35	35	-	-	-	-	-	-
"	22.4	38	35	31	29	28	27	26	26
"	44.8	30	30	26	26	24	24	23	23
"	89.6	29	28	27	25	24	24	23	23
July 31, 1974									
Bermudagrass	0	39	33	32	30	29	28	28	28
"	22.4	38	28	27	25	25	25	25	25
"	44.8	39	29	27	26	26	26	25	25
"	89.6	37	26	25	-	-	-	-	-
September 10, 1974									
Bermudagrass	22.4	25	21	21	20	20	20	20	20
"	44.8	19	19	19	19	19	19	19	19
"	89.6	21	20	20	20	19	20	19	19

*Degrees celsius

TABLE 16. PERCENT WATER, AS DETERMINED BY NEUTRON PROBE, IN LAKEWOOD SAND AT VARIOUS DEPTHS UNDER FOREST VEGETATION WITH (W) AND WITHOUT (W/O) SLUDGE ON FOUR DATES IN THE SECOND SEASON OF APPLICATION

mm depth	7/17/74			7/31/74			9/10/74			10/10/74		
	w/o	w	% Water	w/o	w	% Water	w/o	w	% Water	w/o	w	% Water
152	4.0	8.5	4.5	4.0	8.5	4.5	3.5	6.0	2.0	11.5	11.5	11.5
304	8.5	11.0	12.0	10.0	11.0	12.0	13.5	14.0	10.0	12.5	12.5	12.5
457	14.0	18.0	19.5	15.0	18.0	19.5	16.5	21.5	13.0	17.5	17.5	17.5
610	14.5	21.0	19.5	15.5	21.0	19.5	18.0	23.0	14.0	18.0	18.0	18.0
762	14.0	20.5	20.0	14.0	20.5	20.0	17.0	22.0	12.5	18.5	18.5	18.5
914	15.0	18.0	18.0	15.0	18.0	18.0	17.5	20.0	13.0	17.5	17.5	17.5
1067	16.0	17.5	14.0	16.5	17.5	14.0	17.5	15.5	13.5	13.5	13.5	13.5
1219	14.0	12.0	12.5	14.0	12.0	12.5	16.0	15.5	12.0	10.0	10.0	10.0
1524	13.5	12.0	12.5	14.0	12.0	12.5	17.0	16.0	11.5	9.0	9.0	9.0
1829	12.5	10.5	13.0	14.0	10.5	13.0	17.0	15.5	11.5	-	-	-
2134	10.5	10.5	-	12.5	10.5	-	17.0	-	10.0	-	-	-
2438	9.5	-	-	9.0	-	-	15.0	-	9.5	-	-	-

SURFACE ACCUMULATION OF SLUDGE

The effects of three years application of sewage sludge on the soil surface was measured by means of a soil sampling tube. Depths to sludge-soil interface were measured for each sludge rate on each soil, and in the lateral boom-sprayed portion of each plot as well as the central manifold-sprayed portion. The difference in appearance of the effects of these modes of application can be seen. Equal rates of application were supposed to have been applied by each technique. No striped or row effects could be detected along the plot margins at the time these samples were taken.

The means of ten 2 cm diameter cores from each of the above conditions are given in Table 17. In every comparison the depth of accumulated sludge was less in the central portion of the plots where sludge was sprayed by a manifold than were applied by a boom along lateral edges. When strips were rototilled across plots to a depth of 8 cm, the darker color of a greater accumulation of surface sludge along lateral edges could be readily detected in each plot.

The mean depth of sludge is consistently greater on the plots of Lakewood soil than those on the Woodmansie soil and generally greater than those on the Downer soil. No explanation for this can be offered at this time. The high sludge rate plot (89.6 t/ha/y) on the Lakewood sand was the plot to show highest groundwater contamination with nitrates.

Opportunities for loss by volatilization were greater for the spray mode of application which covered plant and soil surface more completely than did boom application. Hence, greater accumulation of surface crust occurred under the half of each field receiving boom-applied sludge.

Data in Table 18 illustrate the weight of sludge accumulated on the surface of these plots as kg/ha. Note that accumulated sludge ranged from 22% of that applied over the three year period to low rate plots to 13% for the high rate plots. Greater losses of dry matter in high sludge rate plots could be attributed to microbial activity. Conceivably, in the thickest (4.6 cm) crust, aeration and moisture conditions might have been more often optimized for microbial decomposition than in thinner crusts. Leaching and volatilization losses have also been occurring.

QUANTITIES OF ELEMENTS REMAINING IN THE SLUDGE ON THE SOIL SURFACE AFTER 3 YEARS OF APPLICATION

The dried surface crust or cake formed by multiple annual applications of sludge during 1973-75 was sampled June 30, 1976 from each soil series. Samples were analyzed for total content

TABLE 17. DEPTH OF SLUDGE AFTER 3 YEARS OF SLUDGE APPLICATION TO THREE SOILS, BY TWO MODES OF APPLICATION - JUNE 30, 1976

Annual rate, t/ha dry matter	Mode of application	Mean effect of sludge rate						Mean effect of mode
		Downer cm	Lakewood cm	Woodmansie cm				
22.4	boom	1.75*	2.35	2.25			boom = 3.46	
"	spray	1.55	1.65	1.60	1.86			
44.8	boom	3.75	2.75	2.50			spray = 2.58	
"	spray	2.75	2.10	1.75	2.60			
80.6	boom	4.90	5.85	5.05				
"	spray	3.75	4.50	3.55	4.60			
Mean depth of sludge on three soils =		3.08	3.20	2.78				

*Each value within the block is the mean of 10 measurements.

TABLE 18. WEIGHT OF SLUDGE ACCUMULATED ON THE SOIL SURFACE AFTER 3 YEARS OF SLUDGE APPLICATION TO THREE SOILS, BY TWO MODES OF APPLICATION - JUNE 30, 1976

Annual rate, t/ha	dry matter	Mode	Downer	Lukewood	Woodmansie	Fraction		Mean of sludge rate	Mean of Mode
						of that	applied		
			Kg/ha	Kg/ha	Kg/ha	%		Kg/ha	Kg/ha
22.4		boom	13,649	18,382	17,549				
"		spray	12,098	12,868	12,478	22		14,504	
44.8		boom	29,247	21,448	19,498				boom =
"		spray	21,448	16,379	13,649	15		20,278	26,992
89.6		boom	38,140	45,625	39,388				spray =
"		spray	29,247	35,096	27,687	13		35,864	20,106
Mean weight of sludge on soils			23,966	25,966	21,708				

(rather than extractable or available) of 17 elements. The volume of sludge on the soil surface was determined on the basis of sludge depths given in Table 19 and plot size. Weight of sludge remaining on the soil surface was determined from an estimated specific gravity of 0.34, times these volumes.

Of the major fertilizer nutrients remaining in the sludge, N would appear ample for crop production, even at the lowest rate (22.4 t/ha/y) of application, providing 1/5 or more of the N were available to the next crop. Phosphorus accumulated in quantities far exceeding any other mineral element in the sludge and would appear to be adequate for any crop even at low availability rates. Potassium did not accumulate in quantities needed by crops, though it too reflects rates of sludge applied.

The secondary elements, Ca and Mg, were not retained in large quantities in the surface sludge. Soil analyses data elsewhere in this report show that these elements had indeed moved into the upper soil profile.

Iron and aluminum also accumulated in considerable quantities. Heavy metal accumulations do not appear excessive. Availability to plants and solubility and hence leachability of all these elements are pH dependent; at near normal pH, mobility is minimal.

EFFECTS OF SLUDGE ON SOIL PROFILES

Important to the interpretation of the soil profiles is the fact that dolomitic hydrated lime was incorporated into the surface 0-20 cm (as previously described) of the plots planted to Midland bermudagrass, including the controls but not the native vegetation (N.V.) plots.

The pH of the surface 10 cm of mineral soil in the sludge-treated plots decreased annually. Below 10 cm soil pH increased annually to a depth of 50 and possibly 75 cm. Some of the decrease in surface pH may be attributed to leaching of the limestone that was originally applied and incorporated to a depth of 20 cm, but in view of the fact that a high lime sludge accumulated on the surface, the decline of one unit between year 1 and year 3 is appreciable. The limestone incorporated initially would not be expected to evoke an increase of one pH unit in 4 years at the 45 cm depth. The persistence of a high pH in the 10-20 cm depth undoubtedly reflects, in part, the original limestone application.

The effect of rate of sludge application on pH throughout the soil profile does not appear to be consistent. The low pH in the surface 0-10 cm and the high pH in the 10-20 cm depth during the third year of application on Woodmansie sand confirm the trend. The nearly uniform pH of 4.5 throughout the

TABLE 19. KG/HA OF ELEMENTS REMAINING ON THE SURFACE IN THE SLUDGE AFTER 3 YEARS OF APPLICATION

Element	Rates of sludge applied annually								
	Downer Loamy Sand			Lakewood Sand			Woodmansie Sand		
	22.4	44.8	89.6	22.4	44.8	89.6	22.4	44.8	89.6
N	565.00	852.00	1141.00	684.00	830.00	1771.00	659.00	727.00	1471.00
P	2360.00	3562.00	4767.00	2860.00	3467.00	7399.00	2752.00	3038.00	6149.00
K	12.90	22.50	30.20	18.10	21.90	46.80	17.40	19.20	38.90
Ca	304.00	458.00	613.00	368.00	446.00	952.00	354.00	391.00	791.00
Mg	21.60	32.60	43.70	26.20	31.80	67.80	25.20	27.80	56.30
Cu	25.60	38.60	51.60	30.00	37.60	80.20	29.80	32.90	66.60
Zn	60.00	91.00	121.00	72.00	88.00	188.00	70.00	77.00	156.00
Ni	1.00	1.52	2.02	1.22	1.48	3.15	1.17	1.29	2.62
Cd	0.23	0.35	0.46	0.28	0.34	0.72	0.27	0.30	0.60
Cr	1.78	2.68	3.59	2.15	2.61	5.57	2.07	2.29	4.60
Pb	8.30	12.40	16.60	10.00	12.10	25.80	9.60	10.60	21.50
Hg	0.03	0.05	0.07	0.04	0.05	0.11	0.04	0.04	0.09
Al	280.00	442.00	564.00	339.00	411.00	877.00	326.00	360.00	728.00
Mn	7.10	10.70	14.40	8.60	10.40	22.30	8.30	9.10	18.50
Fe	886.00	1337.00	1789.00	1073.00	1301.00	2777.00	1033.00	1140.00	2307.00
B	17.90	27.00	36.10	21.60	26.20	56.00	20.80	23.00	46.60
Na	6.70	10.10	13.60	8.10	9.90	21.10	7.80	8.60	17.50

soil profile on the unlimed plot of native forest vegetation indicates the lack of effect 44.8 kg/ha/y of sludge on soil pH.

All three soils were very similar in soil pH, particularly in that portion of the profile in which limestone was incorporated.

Available soil phosphorus (P) (extractable by North Carolina procedure) increased markedly with increasing rates of sludge applied after three years, to a depth of 15 to 20 cm. The appreciably lower P content in the 44.8 t/ha/y plots in native vegetation (N.V.) as compared with the same rate of sludge applied to bermudagrass may reflect the combined effects of the filtering action of the forest floor litter, and the substantially lower pH of the unlimed soil minimizing P availability.

Availability of P in all sludge-treated plots decreases precipitously with depth to compare closely with the no-sludge plot at a depth of 10-15 cm.

Similarly, a marked surface accumulation in P occurred with time. At the highest rate of application this effect extended 15 cm deep after 3 years of application on Woodmansie sand. As might be expected, sludge applications increased the content of available P of the two sand-textured soils to a greater depth than in the loamy sand (Downer).

There appeared to be no effects of sludge treatment on available soil potassium (K), and this constituent appeared to vary somewhat from year to year without any apparent pattern. Throughout most of the soil profile the fine-textured Downer loamy sand had a slightly higher content of available K (extractable by North Carolina procedure) than did the two sands.

The level of extractable calcium (Ca) appears to be highest in the 10 to 20 cm depth of the lime and sludge-treated plots after three years. No clear effect of rate of application is seen above this depth, but a consistent positive effect continues down to about 50 cm. On a cumulative basis the increase in Ca in the soil profile due to sludge application appears to extend but decline with depth of 135 cm.

Rate of sludge application did not clearly affect the level of available magnesium (Mg). Mg was highest at the 5 to 10 cm depth for all treatments in Lakewood sand after 3 years. Note that the Mg values for 44.8 g/ha/y on the native vegetation (unlimed) were appreciably lower than for the same rate on the limed bermudagrass plot. Much of the Mg in the grass plots therefore came from the dolomitic limestone applied and incorporated in the soil prior to planting the grass in 1973. No cumulative effect or differences between soil series could

be detected.

The mobile nitrate (NO_3) ion showed gross differences between years that most likely reflect temperature and precipitation conditions just prior to sampling. Data after the second year of application showed a generally positive effect of sludge loading rate, particularly in the 0-10 cm depth. The consistently low NO_3 values in the native vegetation plot may relate to the suspension of the sludge above the sampled mineral soil by the surface covering of forest litter. Samples taken after 3 years of sludge application show a slight elevation in NO_3 in the 15 to 75 cm depth. This was probably a temporary effect reflecting particularly heavy loadings of sludge on this plot at the end of the season just prior to sampling.

Values for soil ammonium proved to be variable but low. In aerobic soil ammonia values should be negligible and any positive values may well be an artifact of sample storage.

After the third year of application there was some evidence of a rate-related accumulation of total Kjeldahl nitrogen (TKN) in the surface (0-10 cm) of sludge-treated plots. Other data indicate no differences of importance among soil types. There was, however, a cumulative effect of sludge applications on TKN in the 0 to 10 or 20 cm depths.

The organic matter content of the mineral soils showed little if any increase due to sludge rates applied. The major difference between the control and native vegetation in the 0-10 cm depth is attributed to the fact that the former was cleared by a bulldozer, and the latter had an appreciable accumulation of organic matter above the mineral layer and some of this undoubtedly entered the 0-5 cm portion of the soil profile in ages past.

The cation exchange capacity values for sludge-treated plots were only slightly increased at the 0-5 cm depth above those for control plots. There were no meaningful increases due to rate, or years of application.

Soil conductivity data proved extremely variable. There was a slight trend toward accumulation of soluble salts during the first 2 years of application, but a decrease in the third year. The latter may be attributable to the exceptionally high rainfall in 1975, the third year.

Rate of sludge application increased the DTPA extractable zinc (Zn) in the 0-10 cm depth. The annual increase in available Zn at the high rate of sludge application was significant. Soil series had no appreciable effect on Zn level.

Copper (Cu) like Zn was increased in proportion to rates of sludge applied, particularly in the 0-5 cm depth. More DTPA

extractable Cu was measured from a greater depth (10 cm) where the highest rate of sludge was applied. Similarly a strong cumulative effect can be seen with time, especially in the 0-10 cm depth zone. Values for Cu were similar in all soil series.

Manganese (Mn), like Cu and Zn also increased in the surface soil according to the sludge applications and the number of years sludge was applied, particularly in the upper portion of the soil profile. DTPA extractable Mn was notably higher in the 0-2.5 cm layer of unlimed soil of the native vegetation than at the same sludge rate on limed soil. This is consistent with the reported greater solubility of this element in acid soils.

There was an increase in the DTPA extractable nickel (Ni) in the 0-5 cm depth, due to rate and duration of sludge applications, but at a much lower rate than for Zn, Cu, and Mn. The more acid soils under the native vegetation did not increase the extractable Ni over that of the plot of limed soil receiving a comparable rate of sludge. There was no effect of soil series on extractable Ni.

There was some increase in DTPA extractable lead (Pb) with increasing rate of sludge application, particularly in the 0-10 cm depth. There was an accumulation in the surface layers over the period of years studied. Response to rate of sludge application and soil acidity appear to be of a lower order of magnitude than the four previous elements. Soil series did not differ in extractable Pb.

After 3 years there was no consistent increase in DTPA extractable cadmium (Cd) in the soil profile. There were instances, however, where high Cd values appeared in the surface 0-10 cm. This was unusual and may have been caused by an application of sludge high in Cd.

There were no significant increases in soil chromium (Cr) due to sludge treatments.

Significant changes in extractable elements in the profiles of the three soils studied appears to be related to soil pH. Soil pH was obviously affected both by liming of plots (other than the natural vegetation plots) and by sludge applications.

Analyses of variance of the results of soil test data taken in 1975 were performed on each element previously discussed at ten depths; namely, 0-3, 3-5, 5-8, 8-10, 10-20, 20-30, 30-61, 61-91, 91-122, and 122-152 cm. The three soil series were used as replicates. Significant differences due to rate of sludge application were found for soil pH and six elements at various depths in the soil profile.

Surface soil pH was lower in plots that received high sludge applications than it was in plots with lower application rates or in control plots that receive no sludge. This effect was significant at the 1% level of probability. From 20 to 60 cm the pH of the zero sludge treatment is significantly lower than certain sludge rates. Rate effect is not consistent throughout the whole profile. Sludge appears to have raised the soil pH above that of the no sludge check plots in the 10 to 60 cm depth zone.

Statistically significant differences in extractable P related to the rate of sludge application were found in the upper four increments of the composite soil profile. These increases in soil P were significant only to a depth of 10 cm.

The mean value of extractable calcium (Ca) in the composite profile of the three soils was significantly increased by sludge. Rates of sludge cause consistently proportionate increases in soil Ca concentration only below the 30 cm depth. Soil calcium in plots treated with sludge at this 89.6 t/ha/y rate was significantly higher (at the 5% level of probability) than the no sludge check plot at the greatest depth tested; 122 to 152 cm, indicating appreciable mobility of Ca.

Extractable Zn in the soil profile varies in proportion to the rates that sludge was applied, particularly at the first four depths sampled, to a depth of 10 cm. The increase in Zn at the 30 to 61 cm depth, particularly for the zero sludge treatment represents high native Zn in the Woodmansie sand at this depth in the soil profile.

Each increment of sludge added increased significantly the amount of extractable Cu in three of the upper soil increments. High levels of Cu are not found beneath the plow depth. Some sludge was plowed into the soil the first year.

Extractable Mn in the five surface increments was highly correlated with rates of sludge applied. Below plow depth differences could not be proven.

Similarly, extractable Ni remained primarily in the upper soil strata. As with Zn, the Ni concentration shows higher peak values in the 30 to 60 cm depth for zero sludge treatment than sludge treatments.

SECTION 8

AEROSOLS AND ODORS

MICROBIAL AEROSOLS

The objectives of the microbial aerosol work were the determination of 1) the size of the buffer zone required to insulate surrounding areas from the biological aerosols generated at the sludge disposal sites; and 2) the extent of employee exposure to these aerosols. For reasons discussed later, more limited goals were pursued, namely: detection of changes in plate counts from air samples taken at various distances from the sites 1) before, during, and after spraying, and 2) in the immediate vicinity of the site personnel, in the presence and absence of spraying. The scientific literature was then studied in a partially successful attempt to use these findings to address the original objectives to determine the need for buffer zones and precautions needed to protect employees from exposure to aerosols.

An aerosol consists of liquid and/or solid particles suspended in air. Biological aerosols may then be defined as biological contaminants occurring as solid or liquid particles in the air. The term is usually applied to aerosols containing viable microorganisms, either as individual particles or as constituents of larger ones. Other biological materials of interest, such as pollen or mold spores (when growth is not a concern), are then referred to as aeroallergens. To avoid confusion with other biological materials such as insects and insect parts, arachnids, seeds, and the like, the term microbiological or microbial aerosol will be used here. Viruses, bacteria, fungi, algae, and protozoa may all be found in microbiological aerosols. Of these, algal and protozoan aerosols are generally of less interest than the other three groups.

There is a long and well-established tradition for the use of indicator bacteria in monitoring the public health significance of waters and wastewaters. Unfortunately, no such tradition exists for digested sewage sludge. Use of the same indicators (coliforms, fecal streptococci) is open to question without additional research into their survival rates during digestion, relative to potential pathogens. In assessing microbiological aerosols derived from digested sludge, the problem is even more difficult. Virtually no quantitative data

is available even for the spray disposal of sewage effluents, which has been practiced to a far greater extent. Viruses and sporeforming bacteria and fungi might survive considerably longer than the typical indicator under such conditions. On the other hand, direct enumeration of pathogens is at least as impractical for sludge aerosols as it is for water or wastewater. In fact, "no satisfactory method is now available" for sampling of viruses in aerosols. In view of these considerations, the monitoring method reluctantly decided upon was the use of a "total plate count" similar to that used in the past water and wastewater testing.

All sampling for microbiological aerosols was performed using two Andersen stacked-sieve cascading impactors. This sampler has been recommended as a standard sampling device by the International Aerobiology Symposium.

A major drawback of this sampler for this study is that it is not capable of measuring changes in concentration of viable particles over short periods of time. Rather, it gives a total value for sampling over a given period (usually one minute in our case). For this reason, some efforts were made to utilize an Anderson viable monitor. This device consists of a rotating drum with a nutritive agar surface. During sampling, aerosols are deposited in a continuous spiral line giving information on instantaneous changes in concentration. However, under field conditions, satisfactory results were not obtained and use of this sample was dropped.

On the other hand, the Andersen sampler provided a special advantage over other sampling devices for this study because it automatically segregated particles by size during collection.

The Andersen sampler is designed for use at a sampling rate of 28.31 (1.0 cubic foot) per minute of air. Periodic calibrations of the pumps used in sampling were made to assure maintenance of this rate.

A standard plate count medium, tryptone glucose extract agar (TGE) was used almost exclusively. This medium is capable of supporting the growth of a wide variety of aerobic and facultative bacteria, and some fungi. In a few cases, EMG agar was used as a selective coliform medium. Exactly 27 ml of sterile medium was aseptically placed in each specially standardized glass petri dish. The dishes had aluminum tops in which absorbent pads were placed to prevent the condensation of excess moisture.

After sampling, plates were rushed back to the laboratory (1-3 hr) where they were immediately placed in a 35°C incubator. Incubations were for 20-22 hr, after which colonies were counted with the aid of a magnifying glass.

The major factors of interest for this study were the changes in plate counts with 1) spraying (versus non-spraying, and 2) distance from the site. However, numerous other factors entered into the experimental design: sampling duration, height of sampling, use of different samplers, weather conditions, time and day of sampling, and sampling site. These factors were provided for in the design to the extent possible.

Sampling under the given field conditions and using the total plate count on TGE agar, it was difficult to demonstrate statistical differences between samples.

Microbial aerosol concentrations in the absence of spraying were virtually identical with the background counts observed prior to the first sludge application, averaging 200 per cubic meter (5.7 per cubic foot). This value is consistent with counts reported in the literature. Two exceptional days with very high counts presumably resulted from the sporulation of saprophytes growing on the sludge plots.

At the edge of the plots, microbial aerosol concentration increased during spraying to an average value of 3,000 per cubic meter (86 per cubic foot). After spraying the counts decreased rapidly, but were still elevated compared to the background readings (average, 490 per cubic meter or 14 per cubic foot).

Average concentrations per cubic meter (per cubic foot) during spraying increased to 410 (11.6) at 15m (50 ft) and 190 (5.4) at 30m (100 ft) downwind, but the 30m reading could not be shown to be significantly different than the before sampling "background" of 130 (3.7). Samples collected at these distances had returned to background levels within 5 to 30 minutes after spraying.

Concentrations in samples collected on top of the sludge spraying apparatus, to which the operator was exposed, were similar to those collected during spraying at the edge of the site, averaging 2,600 per cubic meter (73 per cubic foot).

Overall, the percentage of large particles ($\geq 5\mu\text{m}$) increased slightly during spraying (70%) relative to before and after spraying (63% each). Only about 9% of the particles were 1-2mm in diameter, the size range of greatest deposition in the alveoli.

There is a need for the development of a suitable indicator for microbial aerosols produced by spray application of sewage sludge on land.

The potential adverse effects of microbial aerosols produced by the spray applications of sewage sludge on land is difficult to evaluate. Generally, concentrations appear to

diminish rapidly with distance, becoming only marginally higher than background readings at 30m (100 ft) under most conditions. Spores from saprophytes on the sludge plots are still readily discernable at 30m (100 ft), however, (1,700 counts per cubic meter or 47 per cubic foot before spraying, down from 3,900 or 110 at 15m). Thus an adequate buffer zone to insure against exposure to some microbial aerosols must be considerably larger than 30m (100 ft).

ODORS

The major objective of the odor study was to determine the type and extent of odor problems encountered during the disposal of digested sewage sludge by surface spraying on land. Recommended control strategies for such odors were also considered. Odor may be defined as that property of a substance which affects the sense of smell.

While odors are often cited as the major cause of air pollution complaints from the public, in many ways the field of odor measurement and control is a primitive one. For example, although there are several devices (ranging from the extremely simple to the very complex) designed to aid in the measurement of odors, there is generally only one instrument available for making the final analysis -- the human nose. Even the way in which odors are detected by the nose is not firmly established. The observations that similar chemicals may have distinctly different odors, and that very different chemicals may have specific odor may result from the complexity of the problems encountered.

Two basic characteristics of any odor are its quality and quantity (strength). Odor quality may be measured simply by rating it on a scale of hedonic tone or pleasantness/unpleasantness, or it may be described by comparison with other odors. Classification schemes are not uncommon though many difficulties may be associated with their use. One of the more recent schemes includes seven primary odors from which all others are said to be derived. Of course, the ultimate description of odor quality would be listing of the chemical species involved and their relative proportions; this is rarely possible.

The ultimate measure of odor quantity would be to determine the concentrations of the individual odorants present, once they were identified -- although even this could be misleading because of possible interactions among them. At the other extreme, a simple rating scale of strength/weakness can be used. The method most commonly employed involves dilution of the odor to the point at which it is just barely detectable. This point is called the threshold odor concentration. The strength of a given odor sample is then reported as the number of dilutions required to reach the odor threshold.

The ultimate goal of odor control must be the prevention of objectionable odors at the receptor site (i.e., the prevention of air pollution). Achievement of this goal may be approached by preventing (1) the formation of odors; (2) the release or escape of any odors formed; (3) the transmission of odors to a receptor; and (4) the detection of objectionable odors by the receptor.

The greatest successes in odor control are achieved through Approach #1, prevention of odor formation. Approach #4, although sometimes successful, is the least desirable type of control. It involves the use of chemicals to cover up or "mask" the undesirable odor. Because of the subjective nature of odor preference, however, virtually any odor strong enough to act as a masking agent is likely to be objectionable to some people. In rare cases it is possible to find a chemical with an odor which will "counteract" the undesirable odor, causing both odors to disappear. Even when this is possible, it is not necessarily desirable, as it involves intentional addition (at some expense) of chemicals to the air. The potential hazards of various masking and counteracting agents are not well known, but may pose more of a health threat than the odor causing gases.

Odor analysis was first attempted using a standard ASTM dilution method. Hamilton gas-tight syringes, which consist of glass barrels with teflon-sealed plungers and Luer-Lok tips were used. This method proved unsatisfactory.

A backup method involved determination of the maximum distance downwind at which the odor was detectable. It should be noted that this is essentially a dilution method, one in which the natural environmental diffusion process is substituted for controlled dilution in a laboratory. Thus, while the results obtained are less controlled and therefore less precise, they closely represent the type of information desired.

The first measurements of this type were performed by the two individuals doing the preliminary bacteriological aerosol work in the Spring of 1973. In 1975, different 5-6 member panels were assembled to make more detailed observations.

Panelists were also asked to rate the odor on a hedonic scale (pleasantness/unpleasantness) and to describe it, both in their own words and by rating various descriptive words as to their applicability. The first seven descriptive terms used represent the primary odors from classification scheme.

In addition to these specific observations, information was also gathered somewhat more informally by talking with people visiting or working at the sludge disposal site.

Wind speed and direction were measured using a portable wind vane and cup anemometer (Science Associates, Inc., Princeton, N.J.) mounted on a 1.8m (6 ft) pole. A sling psychrometer (#12-7011, Bacharach Instrument Company, Pittsburgh, Pa.) was used to measure air temperature and relative humidity.

Odors at the sludge disposal sites were moderate to slight under most conditions as long as well digested sludge was used.

The maximum distances (panel medians) downwind from the disposal sites at which members of two panels could still detect any odor were 35 and 39m (114 and 127 ft) immediately prior to spraying on two different days. The comparable values during and immediately after spraying were 156 and 219m (718 ft).

The predominant odors at the sites appeared to be derived directly from the sludge, rather than from its decomposition at the site. This was reflected in the odor descriptions, which included stale, earthy, musty, and moldy.

The major emphasis in odor control must be placed on the prevention of odor formation. This includes the necessity for using only well digested sludge and applying it at low enough rate to prevent flooding and anaerobic conditions.

The potential for odor complaints may be further reduced by providing a sufficient buffer zone between the site and any unwilling receptors. A minimum distance of $\frac{1}{2}$ km ($\frac{1}{4}$ mile) would appear desirable.

SECTION 9

GROUNDWATER QUALITY

INTRODUCTION: POLLUTION VS. CONTAMINATION

It is essential prior to evaluation of land disposal, to effectively understand the difference between contamination of a groundwater and pollution of the groundwater.

The following distinction will be used for the discussion in this report. Pollution will be defined in terms of violation of a standard and/or in terms relating to the interference with a desired use of a resource.

Contamination on the other hand relates to an increase in a substance normally present or an addition of a substance not normally present that would neither violate any existing standard nor interfere with the conventional use of the resource.

In the land disposal of an organic sludge, some impact must be expected from the normal biological processes resulting from the further decomposition of the sludge mass. To expect land disposal without some impact on groundwater quality, is not consistent with a knowledge of basic environmental mechanisms which themselves make land disposal a feasible alternative.

Soils are dynamic bodies consisting of three phases - gas, solid and liquid, that are always moving toward equilibrium with the environment. Any manipulation of soils alters the environment and may cause qualitative change in soil water. Some contamination of soil water and usually groundwater is inevitable when soil amendments are applied. Use of sludge as a soil amendment should not contaminate groundwater enough to pollute it and interfere with its desired use. It is imperative, as we discuss groundwater quality violations resulting from land utilization of sludge, that we keep in mind this important distinction between contamination and pollution.

SAMPLING SCHEDULE AND ANALYSIS

Approximately one liter of groundwater was withdrawn and analyzed for as many as 34 different chemical constituents. The 34 chemical constituents analyzed are listed in Table 20.

TABLE 20. GROUNDWATER MONITORING

Total Coliforms	Temperature
Total Dissolved Solids	pH
Ammonia-nitrogen (NH ₄ -N)	Turbidity
Nitrate-nitrogen (NO ₃ -N)	LAS or Alkyl Sulfates
Nitrite-nitrogen (NO ₂ -N)	Specific Conductance
Total Phosphate (PO ₄ -P)	Silica (SO ₂)
Ortho-Phosphate (o-PO ₄ -P)	Potassium (K)
Alkalinity	Sodium (Na)
Hardness	Calcium (Ca)
Total Organic Carbon (TOC)	Sulfate (SO ₄)
Chloride (Cl)	Boron (B)
Fluoride (F)	
Metals (Hg, Zn, Cr, Mn, Fe, Mg, Pb, Cd, Cu, Ni, Al)	

TABLE 21. MODIFIED GROUNDWATER ANALYSIS

<u>pH</u>	<u>Specific Conductance</u>
Total Coliforms	Manganese (Mg)
Temperature	Calcium (Ca)
Total Organic Carbon (TOC)	Hardness
Ortho-Phosphate (o-PO ₄ -P)	Copper (Cu)
Nitrate-nitrogen (NO ₃ -N)	Zinc (Zn)
Chloride (Cl)	Sulfate (SO ₄)

The "comprehensive analysis" of the 34 constituents was performed at intervals of six weeks. At three weeks intervals or after an appreciable rainfall, groundwater samples were also withdrawn for a modified schedule of chemical analyses consisting of 14 determinations set forth in Table 21. A total of 57 groundwater samples were taken from each observation well for comprehensive or modified analyses over a period of three and one-half years.

In order to evaluate sudden changes in groundwater quality, the frequency of groundwater sampling was intensified (beginning in December 1974) to include weekly samples from all observation wells. However, only two (2) chemical constituents were analyzed -- nitrate-nitrogen and specific conductance.

The first sets of profile samples were not installed until July-October 1974. Only occasional samples were taken for analysis of specific conductance and chloride. However, after June 1975, sampling was intensified to study stratified layers of contaminants. The chemical analyses performed on each profile sample included pH, specific conductance, chloride and nitrate-nitrogen.

Although the application of wastewater solids was terminated in November 1975 after three years of solids applications, sampling was continued for an additional year through November 1976. The additional year was necessary to evaluate the impact on groundwater quality for that third and final year of solids application, as well as any cumulative effects that may occur after three years of land spraying of wastewater solids.

Laboratory analyses were performed according to the following references.

- (1) Standard Methods for the Examination of Water and Wastewater, 1971.
- (2) Collection and Analysis of Water Samples for Dissolved Minerals and Gas, U.S. Geological Survey Laboratory Manual, 1970.
- (3) Methods for Chemical Analysis of Water and Waste, Environmental Protection Agency, 1974.
- (4) Mercury Analysis Manual - Flameless Technique, Perkin-Elmer, 1972.
- (5) Chemical Analysis for Water Quality, FWPCA, 1969.

BASELINE GROUNDWATER QUALITY BEFORE WASTEWATER SOLIDS APPLICATIONS

The average concentrations of 33 groundwater chemical constituents are reported in Table 22 for the three (3) individual wastewater solids application sites -- Downer, Lakewood, Woodmansie. Table 22 also includes the range of concentration observed for each constituent. The range and average values reported (Table 22) represent a condensed summary of the data compiled from the chemical analyses of numerous samples withdrawn from observation wells at respective sites.

The groundwater quality before sludge application reported for the Lakewood and Woodmansie sites was similar. At the Downer site the average concentration of most chemical constituents was higher. For example, the total dissolved solids baseline concentrations at the Lakewood and Woodmansie sites was 26 mg/l; however, the average total dissolved solids concentration in groundwater samples withdrawn from the Downer site was 30 mg/l. Most chemical constituents followed this generalization; i.e., chemical constituents in groundwater samples from the Downer site are slightly greater in concentration than those observed at the Lakewood and Woodmansie application sites.

The background groundwater quality before solids applications can be characterized as having a total dissolved solids concentration ranging from 12-82 mg/l (average: 28 mg/l). The groundwater contained only small quantities of common salts -- Cl^- , SO_4^{2-} , Ca^{++} , Mg^{++} , Na^+ , K^+ ; there also appeared to be no coliforms. Total organic carbon levels ranged from 0.5 mg/l to 4.0 mg/l. Low levels of $\text{NO}_3\text{-N}$ were observed varying up to 0.3 mg/l. Only trace concentrations of phosphate were found with one exception; one observation well at the Downer site contained 0.92 mg/l $\text{PO}_4\text{-P}$. In almost all samples only trace concentrations of heavy metals were observed, but a few samples did contain concentrations of metals considerably above the reported average concentration in Table 22: aluminum ranged as high as 2.2 mg/l; copper varied up to 0.057 mg/l; zinc varied up to 0.383 mg/l and nickel was found to be as high as 0.036 mg/l. Lead, chromium, mercury and cadmium were not detected. However, it should be assumed that traces of these metals do exist, but at concentrations below the sensitivity of the analytical procedure.

GROUNDWATER QUALITY AFTER THE APPLICATION OF WASTEWATER SOLIDS

Introduction

To assess the changes in groundwater quality from the land application of wastewater solids, the concentrations of monitored chemical constituents from skim and profile sampling

TABLE 22. AVERAGE BACKGROUND WATER QUALITY BEFORE APPLICATION OF WASTEWATER SOLIDS

Constituent	Colliers Mills Downer		Webbs Mills	
	Lakewood		Woodmansie	
pH	5.0 (4.7 - 5.8)	5.2 (4.9 - 5.6)	5.1 (4.8 - 5.3)	
Total Coliforms	0.5 (0 - 39)	0 (0 - 1)	0 (0 - 4)	
Kjeldahl Nitrogen (mg/l)	0.01 (0.0 - 0.97)	0.001 (0.0 - 0.02)	0.001 (0.0 - 0.10)	
NO ₃ -N (mg/l)	0.01 (0.0 - .3)	0.005 (0.0 - 0.04)	0.008 (0.0 - 0.06)	
NO ₂ -N (mg/l)	0.0	0.0	0.0	
PO ₄ -P (mg/l)	0.0 (Tr)	0.0	0.0	
Total PO ₄ -P (mg/l)	0.008 (0.0 - 0.92)	0.002 (0.0 - 0.02)	0.003 (0.0 - 0.05)	
Chloride (mg/l)	3.9 (1.4 - 8.0)	3.4 (1.8 - 4.8)	3.0 (1.9 - 5.4)	
Specific Conductance (µmhos/cm)	44 (32 - 67)	33 (25 - 76)	33 (30 - 41)	
Total Dissolved Solids (mg/l)	30 (12 - 46)	26 (12 - 82)	26 (21 - 36)	
Total Hardness (mg/l as CaCO ₃)	7.0 (3 - 18)	5.5 (3 - 12)	5.7 (4 - 13)	

TABLE 22. AVERAGE BACKGROUND WATER QUALITY BEFORE APPLICATION OF WASTEWATER SOLIDS
(CONT'D)

Constituent	Colliers Mills		Webbs Mills	
	Downer	Lakewood	Woodmansie	
Sulfate (mg/l)	13 (10 - 26)	10 (8 - 19)	11 (10 - 13)	
Fluoride (mg/l)	0 - 0.01 (0.0 - 0.2)	0.01 (0.0 - 0.2)	10.01 (0.0 - 0.2)	
Total Organic Carbon (mg/l)	1.0 (0.5 - 3.0)	1.1 (0.5 - 4.0)	1.0 (0.5 - 3.0)	
Alkalinity	8 (6 - 10)	7 (6 - 8)	8 (6 - 10)	
Silica (mg/l)	3.2 (1.0 - 4.1)	2.3 (0.9 - 3.7)	2.4 (0.7 - 3.8)	
Iron (mg/l)	0.35 (0.06 - 0.89)	0.30 (0.11 - 1.0)	0.38 (0.11 - 0.58)	
Manganese (mg/l)	0.06 (0.01 - 0.37)	0.02 (0.01 - 0.04)	0.025 (0.02 - 0.05)	
Calcium (mg/l)	2.4 (0.44 - 4.7)	2.1 (0.65 - 4.1)	2.1 (0.89 - 4.1)	
Magnesium (mg/l)	0.42 (0.21 - 1.8)	0.32 (0.18 - 1.4)	0.34 (0.27 - 1.8)	
Sodium (mg/l)	1.8 (0.98 - 3.2)	1.30 (0.98 - 8.0)	1.5 (1.34 - 1.66)	

TABLE 22. AVERAGE BACKGROUND WATER QUALITY BEFORE APPLICATION OF WASTEWATER SOLIDS
(CONT'D)

Constituent	Colliers Mills		Webbs Mills	
	Downer	Lakewood	Woodmansie	
Potassium (mg/l)	0.40 (0.19 - 1.7)	0.35 (0.14 - 3.2)	0.5 (0.17 - 1.6)	
Zinc (mg/l)	0.035 (0.0 - .383)	0.024 (0.0 - 0.125)	0.021 (0.0 - 0.121)	
Copper (mg/l)	.003 (0.0 - 0.057)	.002 (0.0 - .036)	.0015 (0.0 - 0.039)	
Chromium (mg/l)	0.0	0.0	0.0	
Lead (mg/l)	0.0 (Tr)	0.0 (Tr)	0.0 (Tr)	
Nickel (mg/l)	.002 (0.0 - 0.036)	0.0 (0.0 - 0.005)	0.001 (0.0 - 0.025)	
Mercury (mg/l)	0.0	0.0	0.0	
Cadmium (mg/l)	0.0	0.0	0.0	
Aluminum (mg/l)	1.0 (0.2 - 2.2)	0.75 (0.1 - 1.3)	0.8 (0.1 - 1.9)	
ABS	0.0	0.0	0.0	
Turbidity (J.T.U.)	0.0 (0 - 0.8)	0.0 (0.15)	0.0 (0.0 - 2.5)	
Boron (mg/l)	0.0	0.0	0.0	

of each observation well were compared over time to (1) constituent concentrations of control wells, (2) background constituent concentrations and (3) constituent concentrations of other observation wells in the study areas. The comparison was to identify pertinent trends and qualitatively evaluate changes in groundwater quality as related to the following factors:

- (1) Concentrations of sludge related constituents into the groundwater
- (2) Wastewater solids loading rates
- (3) Soil type
- (4) Seasonal variations
- (5) Rainfall
- (6) Cumulative trends over four years
- (7) Nitrate contamination
- (8) Drinking water standards

Analysis of Trends in Groundwater Quality

The results from 34 chemical analyses performed on samples collected over a four-year period (1973-1976) were tabulated to express the range and average chemical constituent concentration observed for all observation wells.

The more mobile anions and cations (Na^+ , Ca^{++} , Mg^{++} , Cl^- , SO_4^{--} and NO_3^-) show measurable groundwater contamination above background and control levels. Heavy metals contamination of the groundwater was generally not detectable, with the possible exception of copper and zinc.

There appears to be no evidence of coliform contamination in any well due to the application of wastewater solids. A few samples did contain one to four coliforms at times inclusive of the control plots. An August 8, 1973 sample from the control well at Colliers Mills - Downer site registered so many coliforms that there were too many to count (TMTC) on the membrane filter.

Upon installation of the wells there was contamination from the solvent used to connect the PVC well casing. After a few flushings of the wells, baseline Total Organic Carbon (TOC) concentrations stabilized between 0.0 and 3.0 mg/l. Total Organic Carbon concentrations in almost all wells were periodically in excess of recorded baseline concentration -- peak concentrations of 24 mg/l and 20 mg/l were recorded for wells,

CM 14B (22.4 t/ha/yr) and WM 21B (44.8 t/ha/yr - natural vegetation) respectively.

Groundwater samples for many of the observation wells showed little or no change in any chemical constituent concentration, but in other wells, peak concentrations of many constituents had increased by more than two (2) orders of magnitude from reported baseline levels. A closer examination will show that in all but a few test wells located outside the plots, the concentration of most chemical constituents analyzed exceeded background levels. The magnitude of contamination above the recorded baseline values was dependent on the loading rate, soil type, and type of vegetation.

The concentration of a chemical constituent within an observation well was dependent on the location of the well with respect to the direction of groundwater flow and whether the wells were located within or downgradient of the application site. Those wells designated as 'A', 'B', and 'C' were within the point of application while the 'D' and 'E' wells were 30.5m (100 ft) and 61m (200 ft) downgradient of the application plots, respectively. While Wells A-C were within an application plot, it would be noted that Well 'A' was installed on the upgradient fringe of the plot, Well 'B' was in the center, and Well 'C' was installed at the far downgradient fringe of the application plot. One would expect little or no change in groundwater quality for Well 'A' because uncontaminated groundwater upgradient of the plot is always flowing through this well. The data seem to support this generalization. Therefore, the evaluation of groundwater quality from the application of wastewater solid (breakthrough) is best interpreted from the data obtained from Wells 'B' and 'C'. Concentrations of chemical constituents in Wells 'B' and 'C' within the same plot can also vary significantly due to the nonhomogeneity of soil; e.g., a clay lens and the boundaries of the contaminate plume.

The 'D' and 'E' wells were installed downgradient of the application site, according to the original water table maps determined at the beginning of the study in 1973. However, it was found that a change in direction of flow occurs under changing conditions of flow gradients, recharge and discharge. The concentration of chemical constituents reported for these wells can tend to become distorted due to changing flow direction. As a result, the values reported may be considerably lower than the representative concentration which lies directly along the mean line of flow. Additional skim samples withdrawn from downgradient wells are also somewhat diluted because these wells are not subject to vertical breakthrough of sludge related chemical constituents, but are still subject to groundwater recharge; hence dilution.

Breakthrough of Sludge-Related Constituents into the Groundwater

The change in chemical concentration beneath the plot specifically refer to the 'B' and 'C' observation wells directly on an application plot. In general, a change in groundwater quality over time was easily seen in the fluctuation of specific conductivity and nitrate-nitrogen in the groundwater. A typical response curve for specific conductance versus time for a contaminated well, WM 13B (Lakewood 89.6 t/ha/y), is shown in Figure 26. Figure 26 shows an increasing stepwise response beginning early in 1974, increasing in 1975 and gradually leveling off during 1976. Figure 27 shows a similar response for nitrate-nitrogen for this well. Additional curves showing cyclic patterns in response to seasonal recharge are shown for WM12B, WM24B (Figure 28) and CM13B, CM13E (Figure 29). Figure 29 also shows the cyclic downgradient movement. The first nitrate peak at well CM13B occurs on April 6, 1976; whereas, the first peak at well CM13E, which is about 300 feet downgradient from CM13B, occurs on October 31, 1975, a time lag of about 7 months (208 days). A calculation based on these data indicate that the contaminant flow velocity is about 1.4 ft/d, which is comparable to velocities of 1.5 ft/d and 1.1 ft/d as determined by other tests at Colliers Mills. The small reduction in nitrate-nitrogen concentration which took place over the distance traveled, may indicate a contaminant slug was maintained for this distance.

The breakthrough of sludge-related constituents above baseline and control concentrations was initially observed at the end of the first year of solids application, November and December 1973. Analysis of groundwater constituents beneath most plots showed an increase in dissolved salt concentrations, particularly nitrate, sulfate, chloride, sodium, calcium and magnesium. There was no breakthrough of coliforms, phosphates and heavy metals.

Contamination occurred in slugs. For example, nitrate-nitrogen breakthrough concentrations of 2.1, 3.52, and 2.5 mg/l occurred on November 6, 1973 in groundwater samples beneath plots WM14, WM13, and WM23 respectively, while the control wells showed a $\text{NO}_3\text{-N}$ concentration of less than 0.1 mg/l. Contaminant concentration had also increased sharply between December 11, 1973 and January 16, 1974, specifically beneath plots WM24, CM12, WM12, CM11, CM13, WM13, and WM23. Nitrate-nitrogen beneath the same plots had also increased from 1.3 to 21.2 mg/l and 1.52 to 18.9 mg/l, respectively. Similar responses were reported for chloride, sulfate, specific conductivity and sodium during this period.

Analysis of groundwater samples taken in 1974 showed contaminant levels had increased above the concentrations reported for late November and December 1973, particularly during the

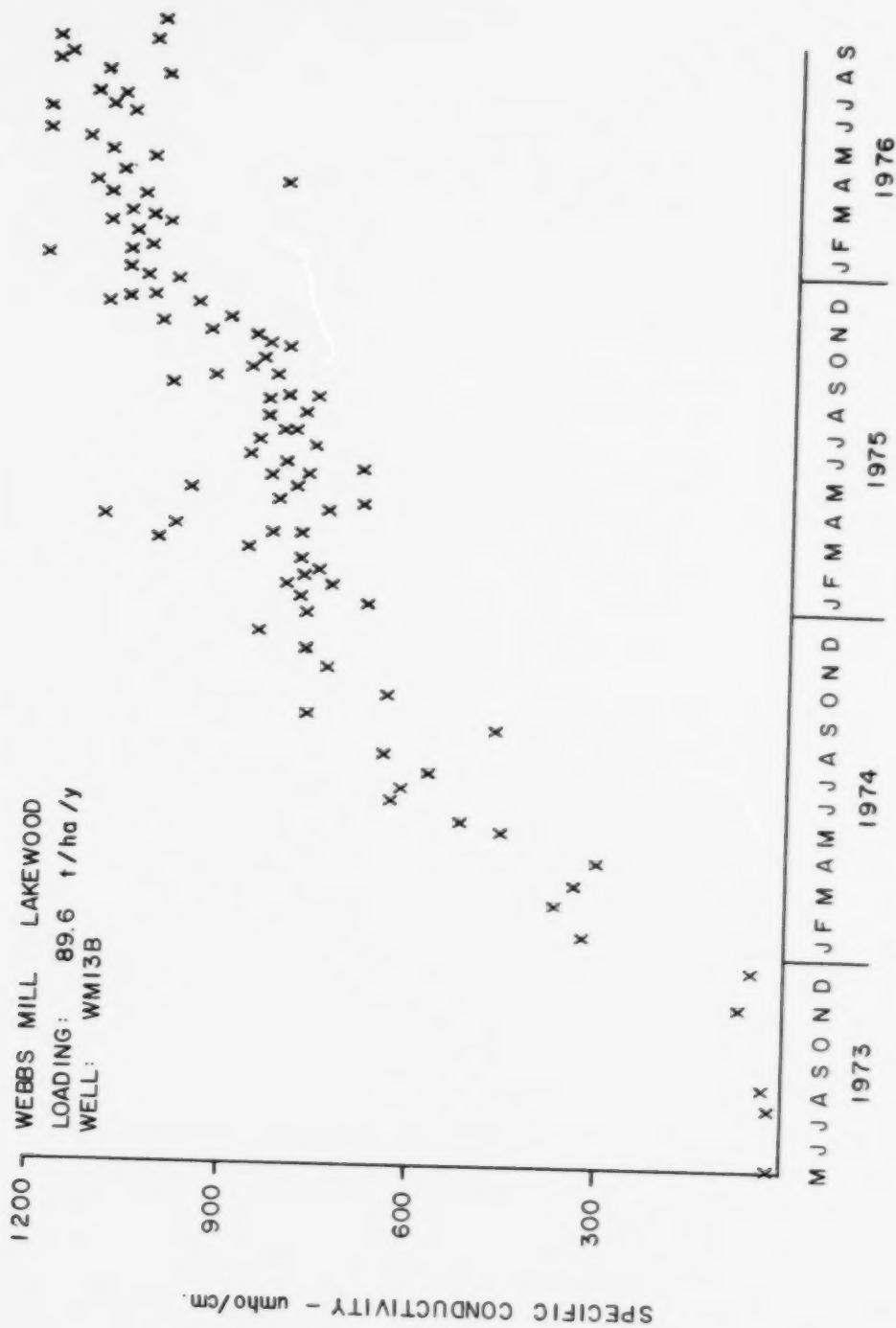


Figure 26. Specific conductance response over time for groundwater monitoring well.

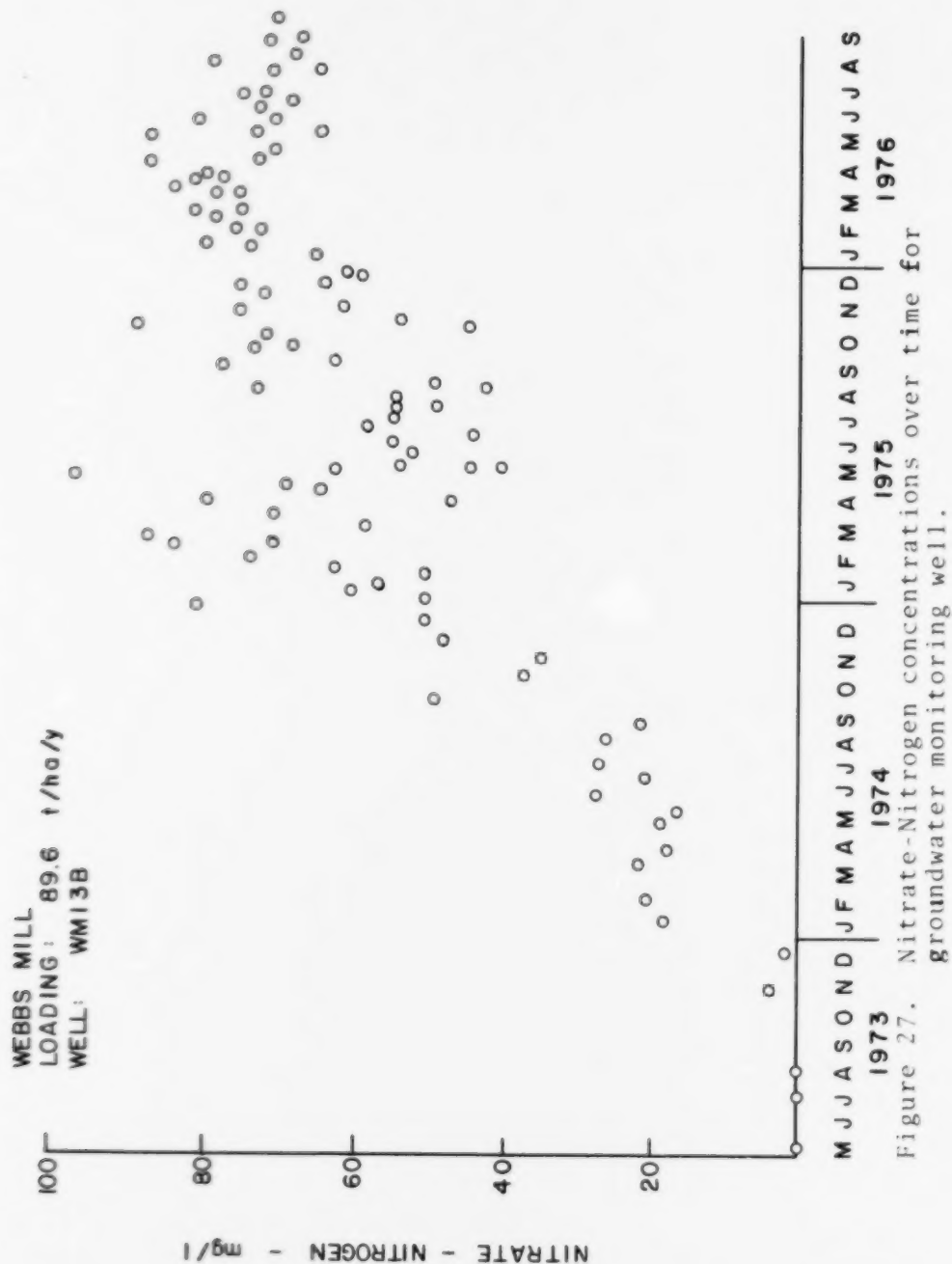


Figure 27. Nitrate-Nitrogen concentrations over time for groundwater monitoring well.

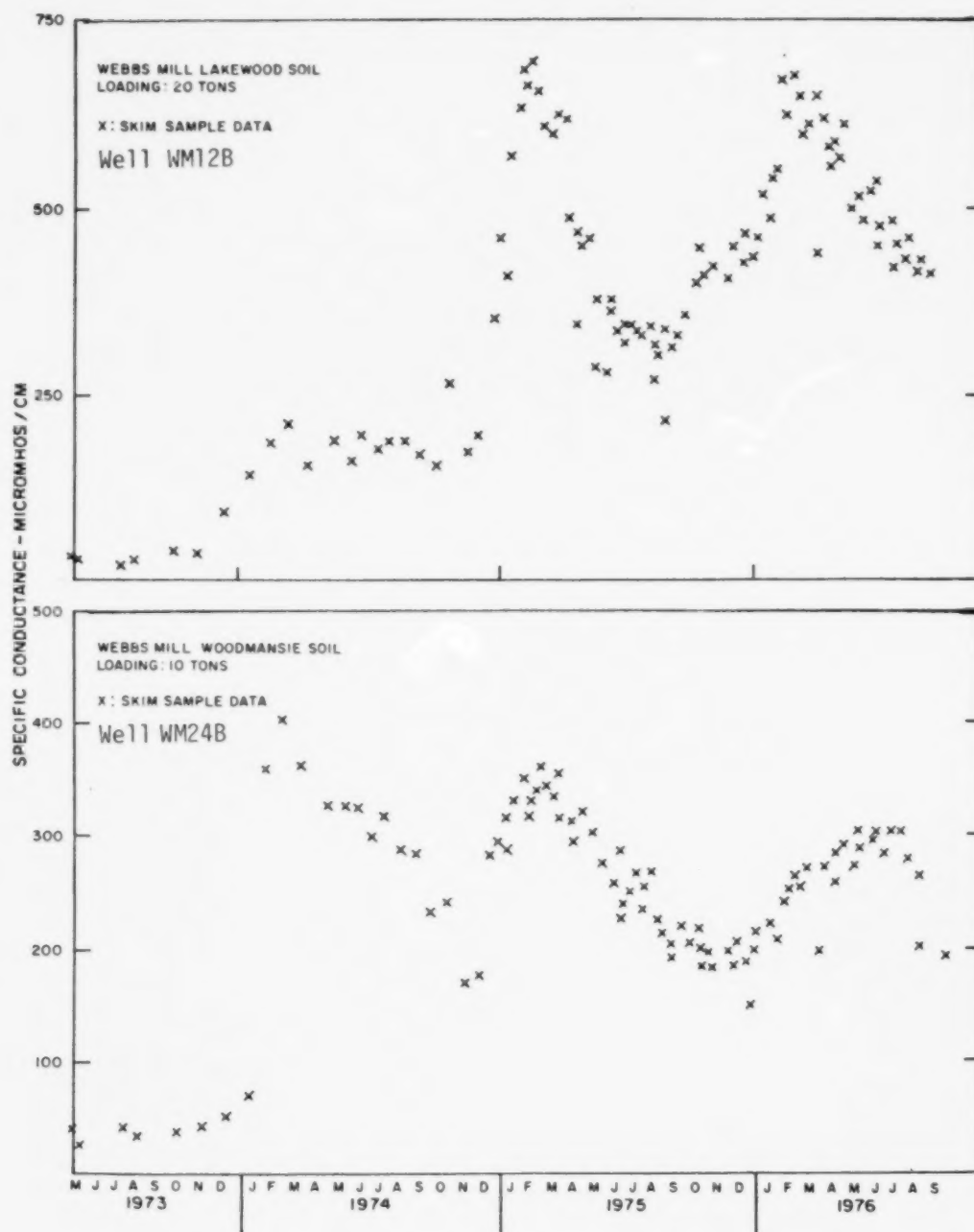


Figure 28. Cyclic fluctuations of specific conductances in groundwater at Webb's Mill.

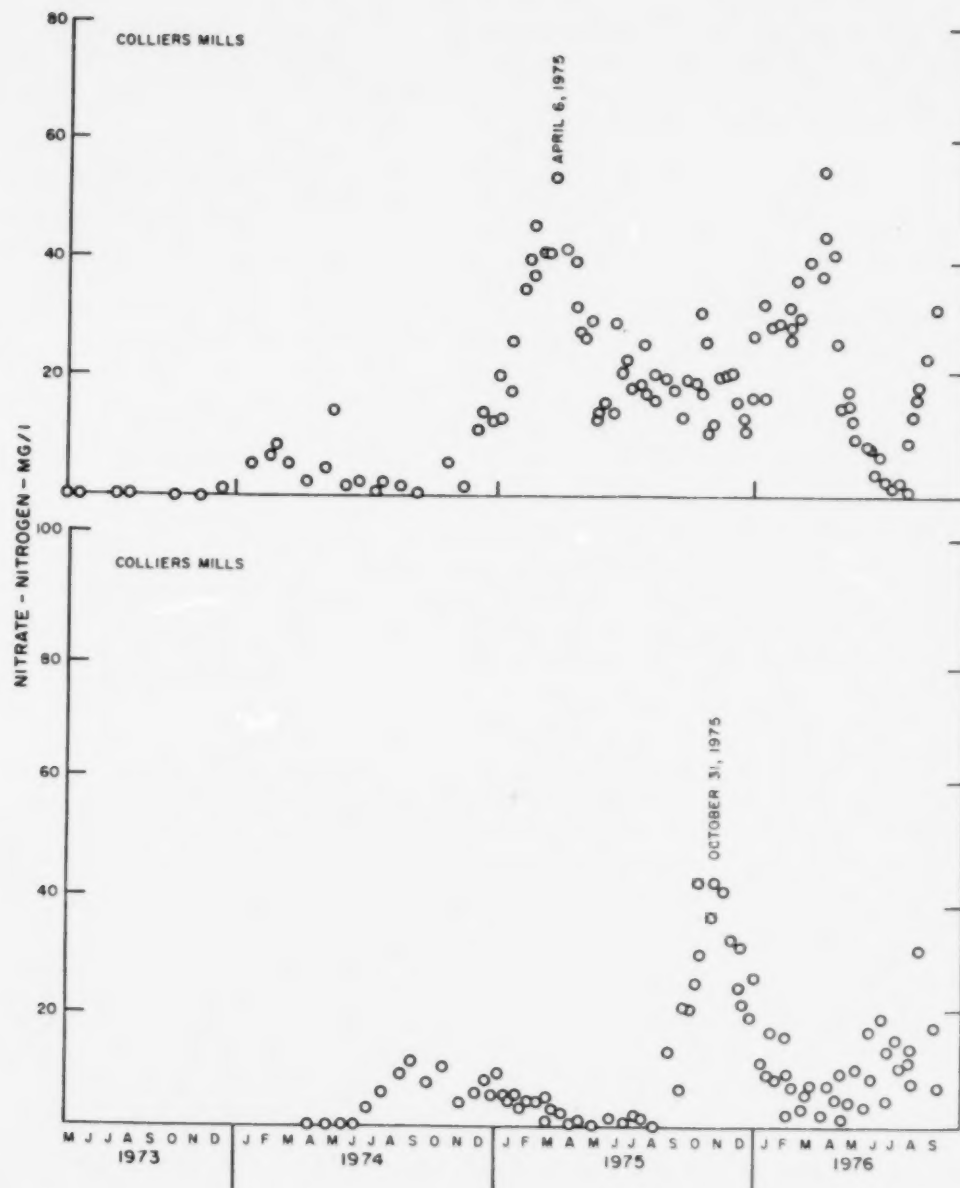


Figure 29. Cyclic fluctuations of Nitrate-Nitrogen at Colliers Mills Wells CM13B and CM13E.

months of January, April, August, October, and late December. The concentrations of the more mobile sludge-related constituents indicate that contamination above background levels had occurred beneath all plots by early 1974; however, contaminant levels at each plot varied according to plot loading rate and soil type. There appeared to be no breakthrough of coliforms, phosphates or heavy metals. In general, the magnitude of contaminant leaching through 1974 appeared to be greatest beneath the following plots: CM13, WM12, CM23, WM21, and WM24.

One of the largest slugs of sludge-related constituents to enter the groundwater occurred in late December 1974. The period between December 1, 1974 and January 19, 1975 indicated a large increase in contaminant concentrations beneath almost all plots, independent of loading or soil type. The 44.8 t/ha/y natural vegetation plots (CM11, WM11, WM21) and the 89.6 t/ha/y plots (CM13, WM13, and WM23) appeared to have the greatest increase in groundwater contamination. For example, beneath the WM21 plot, sulfate concentrations increased from 17 to 133 mg/l, nitrate-nitrogen levels increased from 9.9 to 14.1 mg/l, chloride increased from 21 to 366 mg/l and total hardness levels increased from 8.9 to 172 mg/l (as CaCO₃).

Results for 1975, a record wet year, groundwater samples also revealed that large slugs of contaminants had entered the groundwater during January, April, June, September and October. Contaminant concentrations were also considerably greater than the values reported in 1973 and 1974. Concentrations of the more mobile sludge-related constituents were above baseline and control levels beneath all plots, while breakthroughs of phosphate and coliforms were not found. Concentrations of heavy metals, especially copper and zinc, also showed a slight increase above baseline levels beneath the WM13 (Lakewood - 89.6 t/ha/y) and WM21 (Woodmansie - 44.8 t/ha/y) plots.

Of particular interest is the contaminant breakthrough that occurred during September and October 1975, a significant recharge period. The sudden increases in nitrate-nitrogen, specific conductance, chloride, sodium and total hardness from September 19, 1975 to October 31, 1975 indicate the transport of a large slug of contaminants into the groundwater. The data indicate that there was little contaminant concentrations change beneath the 22.4 t/ha/y plots during the period, while plots receiving a 44.8 t/ha/y and 89.6 t/ha/y had a very large increase in concentration of many sludge-related constituents. The sulfate concentrations increased from 40 to 100 mg/l beneath the CM12 plot, from 70 to 200 mg/l beneath the WM21 plot, and from 140 to 250 mg/l beneath the CM13 plot. Nitrate-nitrogen concentrations also increased sharply beneath these plots, 33.6 to 66 mg/l, 22.3 to 65 mg/l, and 11.6 to 84 mg/l, respectively.

Leaching occurred in all plots in January, February and April of 1976. The average contaminant concentration was generally equivalent or less than the levels reported in 1975; but, the average contaminant concentrations did appear to increase beneath the Lakewood WM12 (44.8 t/ha/y) and WM13 (89.6 t/ha/y) plots. There was no breakthrough of phosphates or coliforms; however, heavy metal concentrations, specifically copper and zinc, were again above background levels beneath the WM13 (Lakewood - 89.6 t/ha/y) and WM21 (Woodmansie - natural vegetation).

The vertical distribution of nitrate-nitrogen may illustrate the nitrate-nitrogen buildup in groundwater from sludge application. The highest concentration of nitrate-nitrogen at the 89.6 t/ha/y plot did not occur at the water table, but at a depth within the plume. Nitrate-nitrogen concentration ranges from about 100 mg/l within the plume to less than 10 mg/l at a depth of about 20 feet below the water table.

The vertical nitrate-nitrogen concentration distribution resulting from a 44.8 t/ha/y application rate on August 2, 1976 ranged from about 36 mg/l at the water table to less than 1.0 mg/l at a depth of about 9 feet below the water table. The nitrate-nitrogen concentration distribution resulting from a 22.4 t/ha/y application rate on September 24, 1976 ranged from about 20 mg/l within the plume to about 0.10 mg/l at a depth of about 8 feet below the water table.

The difference in vertical distribution pattern, where the higher concentration of contaminants are at the water table on August 2 and below the water table in September and October can be attributed to the cyclic pattern of movement of the contaminants in response to recharge. On August 2, 1976, the trough of the cycle had not been reached and the higher contaminant concentration was still at the water table. By September 24, 1976 the low point of the cycle had occurred and the higher contaminant concentration had moved deeper into the saturated zone.

In the next recharge period in the late fall and winter of 1976 and early 1977, the cyclic change in the concentration of nitrate-nitrogen should again occur.

Effects of Loading Rates on Groundwater Contamination

As discussed in previous sections, each site consisted of five plots; three plots were cleared of all natural vegetation, in order to plant Midland bermudagrass and cool season rye grass. Another plot remained in its natural state, while the fifth plot served as a "control". The wastewater solids loading rates selected were as follows:

- (1) 22.4 t/ha/y
- (2) 44.8 t/ha/y
- (3) 44.8 t/ha/y - natural vegetation plot
- (4) 89.6 t/ha/y
- (5) Control - no wastewater solids applications

Groundwater contamination was observed beneath all plots receiving wastewater solids applications. Contaminant concentrations of most sludge-related constituents generally increased with increasing loading rates beneath the cleared plots. This was particularly true at the Colliers Mills-Downer and Webbs Mill-Lakewood locations.

It should be noted that this discussion pertaining to the 44.8 t/ha/y wastewater solids application rate is limited to the three cleared 44.8 t/ha/y plots. The 44.8 t/ha/y natural vegetation plots were not discussed in this section because plot characteristics were significantly different than that of a cleared plot. The cleared plots were limed, the natural organic layer was removed, and cover crops were planted and harvested, while the natural vegetation plot remained in its "natural state".

Although groundwater contamination increased with increasing loading rates at the Downer and Lakewood sites, groundwater quality at the Webbs Mill site on Woodmansie soils did not substantiate this trend. Groundwater contamination was greater at a wastewater solids loading rate of 22.4 t/ha/y than that of the larger 44.8 t/ha/y loading rate. Average contaminant levels beneath the 22.4 t/ha/y were almost equivalent to the results reported at the Woodmansie 89.6 t/ha/y. These observations would suggest a 22.4 t/ha/y wastewater solids loading could have more adverse impact on groundwater quality than the larger 44.8 t/ha/y solids loading. Although groundwater quality data at the Woodmansie location support this anomalous observation, it can be explained by the non-homogeneity of the soil. The original well driller's log indicates a possible pocket of sand on the Woodmansie 22.4 t/ha/y plot in the area of the 'B' well. The sand pocket is not characteristic of a Woodmansie soil, which has more silt and clay development. However, the sandy soil pocket would account for the large peak and average contaminant breakthrough concentrations observed on this 22.4 t/ha/y plot. Therefore, the groundwater quality data reported for the 'B' well on the Woodmansie 22.4 t/ha/y plot could be considered "atypical" of a Woodmansie soil and was not included in the development of any conclusions pertaining to the Woodmansie soil.

Groundwater contamination was observed beneath all cleared application plots, independent of loading rates. However, the magnitude of groundwater contamination above back-

ground levels for most sludge-related constituents generally increase proportionally with increasing wastewater solids loadings.

Effects of Soil Type on Transmission of Contaminants into Groundwater

Wastewater solids were distributed to three separate application sites, each having a different soil - Downer series, Lakewood series, and Woodmansie series.

It should be noted that the following discussion pertains only to the cleared plots. No reference was made to the natural vegetation plots because soil conditions are significantly different. The natural vegetation plot was retained at its natural organic layer and received no lime additions to control soil pH.

Groundwater contamination was observed initially at the Lakewood Sand location. Contaminant concentrations were also greatest at this site. The Lakewood 89.6 t/ha/y plot had an average contaminant concentration 1.5 to 4.0 times greater than the average contaminant levels observed beneath the 89.6 t/ha/y Woodmansie and Downer plots for such sludge-related constituents as nitrate-nitrogen, specific conductivity, chloride, sulfate, sodium, and calcium. Average contaminant breakthrough concentrations beneath the Lakewood 44.8 t/ha/y plot were also 1.5 to 4.0 times greater than the average levels reported at the Downer and Woodmansie plots receiving the same wastewater solids loadings.

Contaminant concentrations at the Downer 44.8 t/ha/y plot were slightly greater than the contaminant level found beneath the Woodmansie 44.8 t/ha/y plot.

The Downer and Lakewood 22.4 t/ha/y plots showed no significant differences in average concentration for nearly all constituents analyzed. Breakthrough concentrations beneath the Woodmansie 22.4 t/ha/y plot were generally two to three times greater than those found at the Downer and Lakewood locations. It should be noted that the groundwater quality results for the Woodmansie 22.4 t/ha/y plot were obtained from observation Well 'B' (center well). Comparison of groundwater quality data for the 'B' and 'C' wells at each 22.4 t/ha/y plot tend to indicate contaminant levels were similar at Downer and Lakewood sites, but significantly different for the Woodmansie sand. For example, average nitrate-nitrogen concentrations (Figure 30) for the 'B' and 'C' wells at the three 22.4 t/ha/y application plots were quite similar at the Lakewood and Downer soils, however, average NO₃-N levels beneath Woodmansie soil varied significantly: 11.9 mg/l for the 'B' well and 4.4 for the 'C' well. The

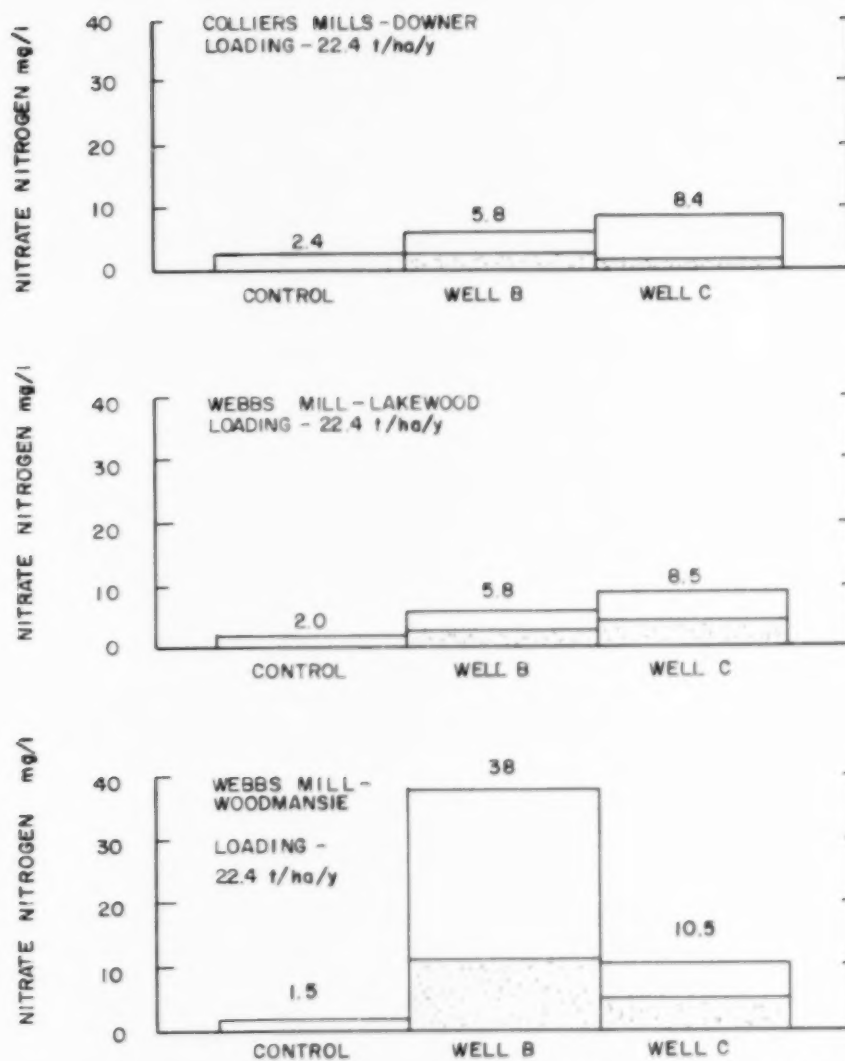


Figure 30. Peak and average Nitrate-Nitrogen break-through concentration for "B" and "C" Well beneath 22.4 t/ha/y Plot.

peak $\text{NO}_3\text{-N}$ concentrations were also disproportionate: 38 mg/l and 10.5 mg/l for the 'B' and 'C' wells, respectively. The well driller log indicated that the soil characteristics of Well 'C' at the Woodmansie 22.4 t/ha/y application was mostly sand. However, soil characteristics in the area of Well 'A' and 'C' indicate considerable silt and clay development. Therefore, breakthrough concentration of contaminants would be considered higher at Well 'C'. Considering the soil characteristics at the Well 'B' emplacement, the groundwater quality data should be considered not representative of the Woodmansie soil location.

The application of wastewater solids on the Lakewood soils appeared to produce the greatest breakthrough of contaminants into the groundwater. The Woodmansie soil had the lowest contaminant concentrations because of considerable silt and clay development in the lower face and substratum of the soil. Where soil characteristics were similar to that of a coarse sand and substratum clay development was minimal, even the low wastewater solids loading rate of 22.4 t/ha/y can cause high contaminant breakthrough concentrations. Contaminant leaching according to soil type and solids loading rates were as follows:

Loading Rate t/ha/y	Contaminant Breakthrough According to Soil Series
22.4	Lakewood > Downer > Woodmansie
44.8	Lakewood >> Downer >> Woodmansie
89.6	Lakewood >>> Downer >>> Woodmansie

It should be noted that the above conclusions do not reflect the groundwater quality results reported for well 'B' on the 22.4 Woodmansie application plot because the well was installed in a sand not characteristic of the Woodmansie soil series. This particular well had contaminant breakthrough concentrations almost equivalent to the levels observed beneath the Woodmansie 89.6 t/ha/y plot.

GROUNDWATER CONTAMINATION BENEATH THE NATURAL VEGETATION PLOTS

At each application site one plot was retained in its "natural" state, to assess the effects of surface spraying of wastewater solids on a natural vegetation-soil-groundwater system. A wastewater solids application rate of 44.8 t/ha/y was utilized on these plots. Unlike the cleared plots where the organic layer was removed and the soil pH was adjusted, the natural vegetation plots were not disturbed and no additional lime was applied to control pH.

The breakthrough of sludge-related constituents was in general not observed during the first year (1973) of groundwater sampling. However, analyses of samples obtained in January and April 1974 did show that concentrations of many groundwater constituents had increased above background levels. For example, nitrate-nitrogen levels increased from an average background of less than 0.2 mg/l, to 3.0 and 6.0 mg/l beneath the Downer, Lakewood, and Woodmansie natural vegetation plots respectively. After April 1974 contaminant concentrations decreased until seasonal recharge occurred December 1974. In December contaminant breakthrough concentration of most sludge-related constituents more than doubled the values reported that previous April. Table 23 shows the adverse effect on groundwater quality caused by this December slug of contamination. For example, the chloride concentration increased to 366 mg/l beneath the Woodmansie natural vegetation plot.

TABLE 23. CONTAMINANT CONCENTRATION BENEATH THE NATURAL
VEGETATION PLOTS

Contaminant	Plct		
	Colliers Mills Downer	Lakewood	Webbs Mill Woodmansie
April 5, 1974			
Specific Con- ductivity- umhos/cm	107	137	265
Nitrate- nitrogen- mg/l	3.2	3.0	6.0
Chloride mg/l	22	17	24
Sulfate- mg/l	24	10	48
Total Hard- ness-mg/l	11.8	15.5	37
December 22, 1974			
Specific Con- ductivity- umhos/cm	340	316	796
Nitrate- nitrogen- mg/l	16.5	17.2	14.8
Chloride- mg/l	68	52	366
Sulfate- mg/l	46	36	133
Total Hard- ness-mg/l	50	63	172

In 1975 average contaminant concentrations beneath the Downer and Woodmansie natural vegetation plots increased sharply above those reported in 1974. At the Lakewood natural vegetation site, contaminant concentrations were generally lower with only a periodic slug of contamination entering the groundwater.

Analysis of the 1976 groundwater quality data indicated contaminant concentrations similar to the levels reported in 1975 for the first four months of the year. However, after April contaminant levels began to decrease considerably, but concentrations beneath these plots did not return to natural background levels.

The general changes in groundwater quality as a result of contamination from wastewater solids applications is summarized by the response of $\text{NO}_3\text{-N}$ over time shown in Figures 31, 32 and 33 for the Downer, Lakewood, and Woodmansie natural vegetation plots. Figure 31 for the Downer Plot and Figure 33 for the Woodmansie plot identify the initial contaminant breakthrough in early 1974, the large increase in contaminant concentrations in 1975 and early 1976, and the gradual decrease in contaminant levels during the Spring and Summer of 1976. However, the magnitude of contaminant concentrations is quite different. The difference in contaminant levels is even more evident when compared to the contaminant response at the Lakewood site (Figure 32). The initial breakthrough of contaminants is similar to that of the Downer and Woodmansie on natural vegetation site, but after the initial breakthrough only an occasional slug of contamination enters the groundwater. The marked differences in contaminant levels seem to be caused by differences in soil pH. The Woodmansie soil had the lowest pH, followed by the Downer soil and Lakewood soil, respectively. The range and average soil pH beneath the natural vegetation plot for the three soil types are shown in Table 24.

TABLE 24. SOIL pH BENEATH NATURAL VEGETATION PLOTS

<u>Soil Type</u>	<u>pH</u>	
	<u>Range</u>	<u>Average</u>
Woodmansie	4.0 - 4.6	4.3
Downer	4.0 - 4.7	4.7
Lakewood	4.0 - 6.8	5.0

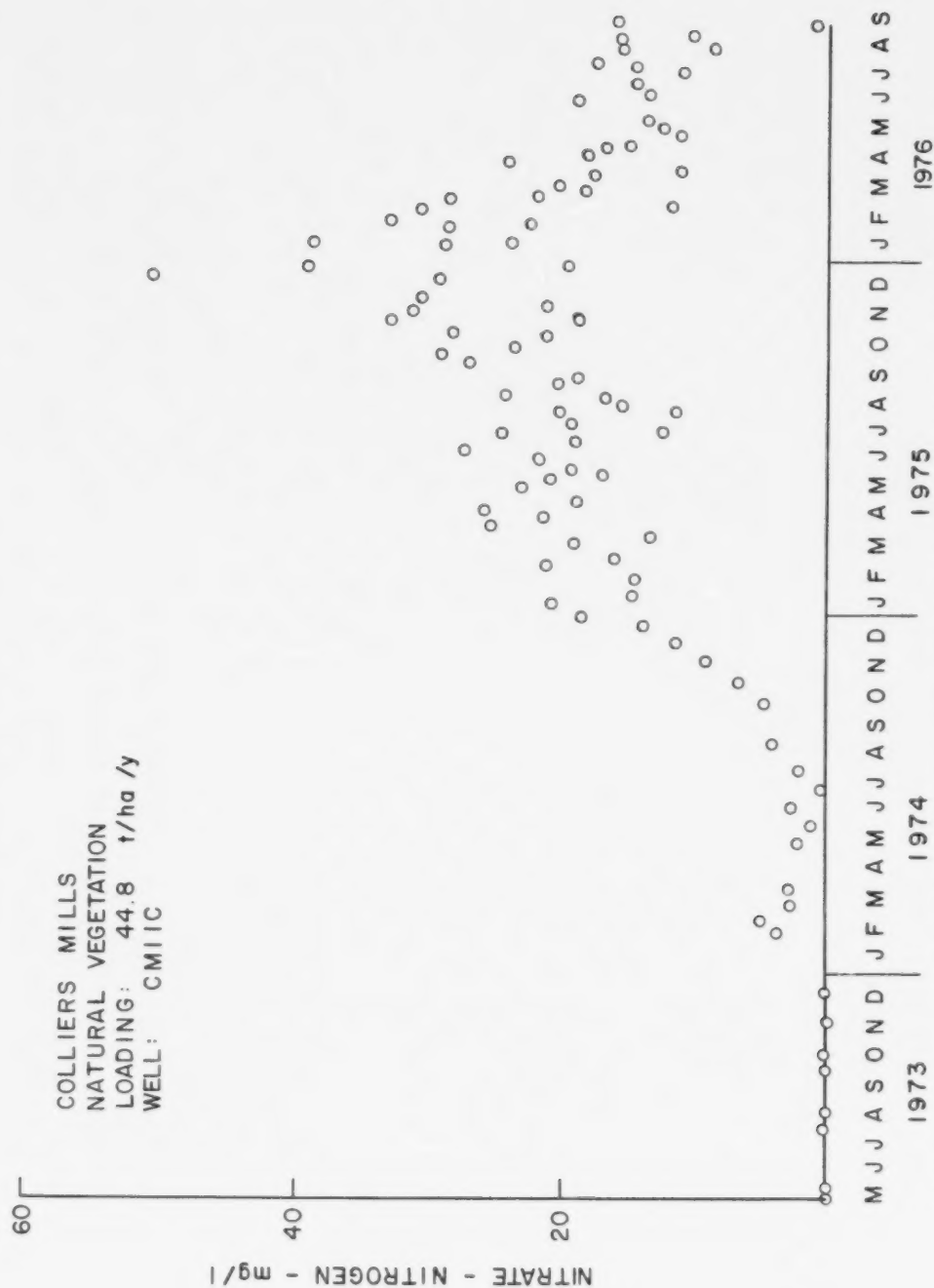


Figure 31. Nitrate-Nitrogen concentration over time for groundwater monitoring well in Downer Loamy Sand.

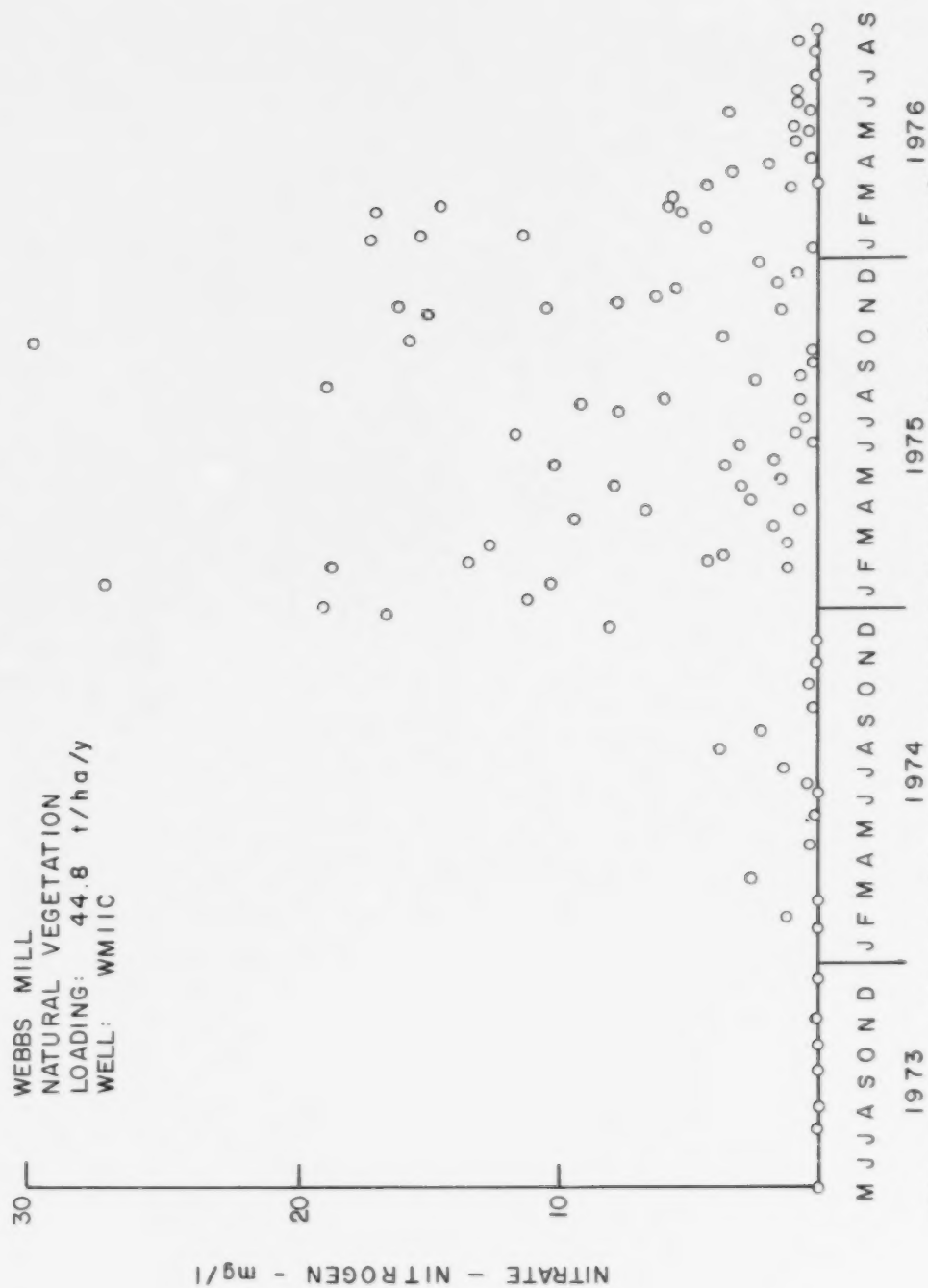


Figure 32. Nitrate-Nitrogen concentration over time for groundwater monitoring well in Lakewood Sand.

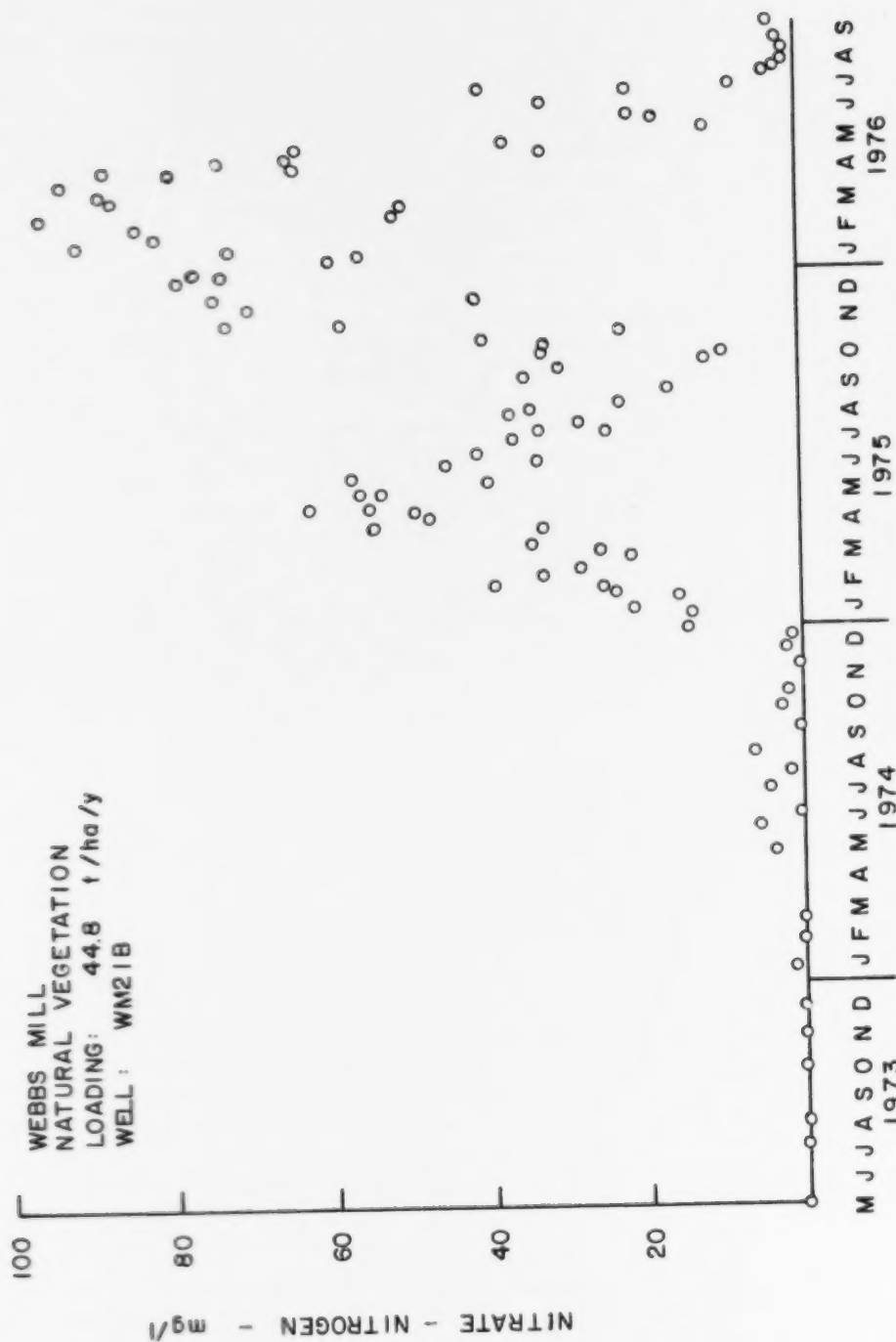


Figure 33. Nitrate-Nitrogen concentration over time for groundwater monitoring well in Woodmansie Loamy Sand.

The depressed pH of the Downer and Woodmansie soils appears to explain the greater leaching of sludge-related constituents observed at these natural vegetation plots as compared to the contaminant concentrations observed beneath the Lakewood natural vegetation plot. According to soil pH contaminant concentration should be greatest at the Woodmansie site, second at the Downer site, and lowest at the Lakewood site. The average concentrations of specific conductivity, nitrate-nitrogen, chlorides, sulfates, sodium, calcium, ammonia nitrogen, and total organic carbon tend to support this argument.

The applications of wastewater solids on natural vegetation plots caused a significant deterioration of groundwater quality. The naturally acid soil conditions which were not corrected by the addition of lime resulted in concentration of many sludge-related constituents approaching and even exceeding the values reported for the more heavily loaded 89.6 t/ha/y application plots. Contaminant concentration beneath the three natural vegetation plots seem to vary according to the soil pH. Groundwater contamination was the greatest beneath the Woodmansie plot, where the average soil pH was as low as 4.3. These acid conditions significantly increased the mobility of many sludge constituents, particularly ammonia nitrogen, organic acids (TOC) and even heavy metals. Compounding the acid soil conditions is the lack of harvesting of a cover crop which would remove many of the sludge constituents from the application site. Therefore, the natural vegetation site becomes a permanent repository for wastewater solids, thus increasing the potential for groundwater contamination.

From these observations it appears the application of wastewater solids at a rate of 44.8 t/ha/y is not suitable for natural vegetation areas typical of the New Jersey Pine Barrens.

Cumulative Effect on Groundwater Quality from Yearly Application of Wastewater Solids

The cumulative impact of wastewater solids applications on groundwater quality was studied over a four (4) year period. Application of wastewater solids in Ocean County was initiated in May 1973 and was applied for three (3) years until October 1975. There were no solids applied during the late fall and winter months. Groundwater monitoring was also performed during this period and was continued for an additional year through October 1976, thus a complete year of groundwater quality data was obtained after wastewater solids applications were discontinued.

Contaminant breakthrough concentrations beneath the 22.4 t/ha/y application plots initially increase, but then stabilized after two years of wastewater solids applications. At the other

extreme, contaminant levels beneath the sandy soil Lakewood 44.8 t/ha/y and 89.6 t/ha/y plots steadily increased over the four-year study. Contaminant concentrations also increased beneath the Woodmansie 44.8 t/ha/y during all four years of the project, however, the increase was very small and gradual as compared to the breakthrough concentrations observed at the Lakewood plot receiving the same wastewater solids loading. Beneath all other plots, contaminant concentrations increased during the first three years of the study, but decreased or remained stable during the fourth and final year of sampling. Although a decrease in contaminant levels was observed at many plots, in no case did the contaminant concentrations of such sludge-related constituents as chloride, sulfate, calcium and nitrate return to background levels. Table 25 summarizes these relative yearly changes in contaminant breakthrough concentrations for the 12 wastewater solids application plots.

Effects of Rainfall on Contaminant Breakthrough

The downward displacement of sludge-related constituents in the soil is dependent upon the frequency and intensity of rainfall and time of the year. Late spring and summer months are a period having high evapotranspiration rates and maximum uptake of sludge constituents by cover crops. However, periodic slugs of contamination were observed during the summer months as a result of very frequent and heavy rains, particularly in 1975. The 1975 rainfall was considerably above the normal for this area. The fall and winter months can be characterized as seasons with low evapotranspiration rates and minimal plant uptake of sludge constituents, thus transport of water into the groundwater from resulting rainfalls was increased and in turn, groundwater contamination sharply increased.

As the intensity of a rainfall increased contaminant breakthrough concentrations were found to increase. It was also observed that the longer the dry period previous to a rainfall, the greater the contaminant breakthrough levels beneath the application plots following the rainstorm. The critical periods of contaminant breakthrough appeared to occur during rainfall events which recharged the aquifer.

A single rainfall event cannot be correlated to the breakthrough of sludge-related constituents entering the groundwater. Other factors must be considered concurrently when analyzing contaminant breakthrough. The factors are the intensity of rainfall, frequency of rainfall, dry period previous to a rainfall event, evapotranspiration rate, growing season, frozen ground and rainfall events leading to the direct recharge of the aquifer. Perhaps the best parameter associated with the occurrence of contaminant breakthrough is recharge

TABLE 25. RELATIVE YEARLY CHANGES IN CONTAMINANT CONCENTRATIONS BENEATH INDIVIDUAL APPLICATION PLOTS FROM 1973 TO 1976

Location	Loading (t/ha/y)	Year			
		1973	1974	1975	1976
Colliers Mill Downer	22.4	slight increase	slight increase	stable	stable
	44.8	slight increase	increase	slight increase	decrease
	44.8#	slight increase	increase	increase	stable
	89.6	slight increase	increase	increase	decrease
Webbs Mill Lakewood	22.4	slight increase	slight increase	stable	stable
	44.8	slight increase	increase	large increase	increase
	44.8#	slight increase	slight increase	slight increase	slight decrease
	89.6	slight increase	large increase	increase	increase
Webbs Mill Woodmansie	22.4	(B) slight increase	slight increase	stable	stable
	(C)	slight increase	increase	stable	stable
	44.8	slight increase	slight increase	slight increase	slight increase
	44.8#	slight increase	increase	large increase	large decrease
	89.6	slight increase	increase	increase	decrease

Natural Vegetation

(B), (C) Well Designation for Well 'B' and 'C' on 22.4 t/ha/y Woodmansie plot.

Stable: No significant change in groundwater quality from previous year.

events. A rainfall or series of rainfalls resulting in groundwater recharge, can considerably increase contaminant concentration levels beneath the application plot; however, successive recharge events can also cause a dilution in contaminant concentration beneath the plot, near the water table. Therefore, yearly analysis of well hydrograph should be considered in conjunction with groundwater quality monitoring to identify periods of contaminant breakthroughs. Generally, the late fall and winter months, when the aquifer is recharging, is when the application plots are most susceptible to contaminant breakthroughs. This behavior is illustrated in Figures 34 and 35 in which monthly response of specific conductance and nitrate-nitrogen are compared to fluctuations in groundwater table and rainfall events.

Seasonal Variations in Contaminant Concentrations

Breakthrough concentrations beneath an application plot were considerably greater during the fall and winter months. Sludge-related constituents are generally more mobile as a result of increased moisture levels in the soil. The rates of reversion and evapotranspiration also are substantially lower. Climate and soil conditions are optimum for groundwater recharge. Since there is no active growth during these months, sludge-related constituents normally utilized by the cover crops are also transported through the soil solution into the groundwater. Therefore, rainfall during the fall and winter months which recharged the aquifer caused considerable groundwater contamination.

Figure 36 clearly illustrates these seasonal differences in contaminant breakthrough beneath an application plot.

Comparison of Contaminant Concentrations to Drinking Water Standards

Contaminant breakthrough concentrations were compared to National Intern Primary Drinking Water Regulations proposed by the U.S. Environmental Protection Agency and the Potable Water Standard established by the State Department of Environmental Protection of the State of New Jersey.

These standards were compared to the average increase of contaminant breakthrough concentrations over background levels at each application plot. A total of 23 groundwater constituents were tabulated according to plot loading rates and soil type. Table 26 presents the average increase of contaminant breakthrough concentrations for these 23 constituents. It should be noted that there are groundwater quality data for two observation wells for the Webbs Mill, Woodmansie 22.4 t/ha/y

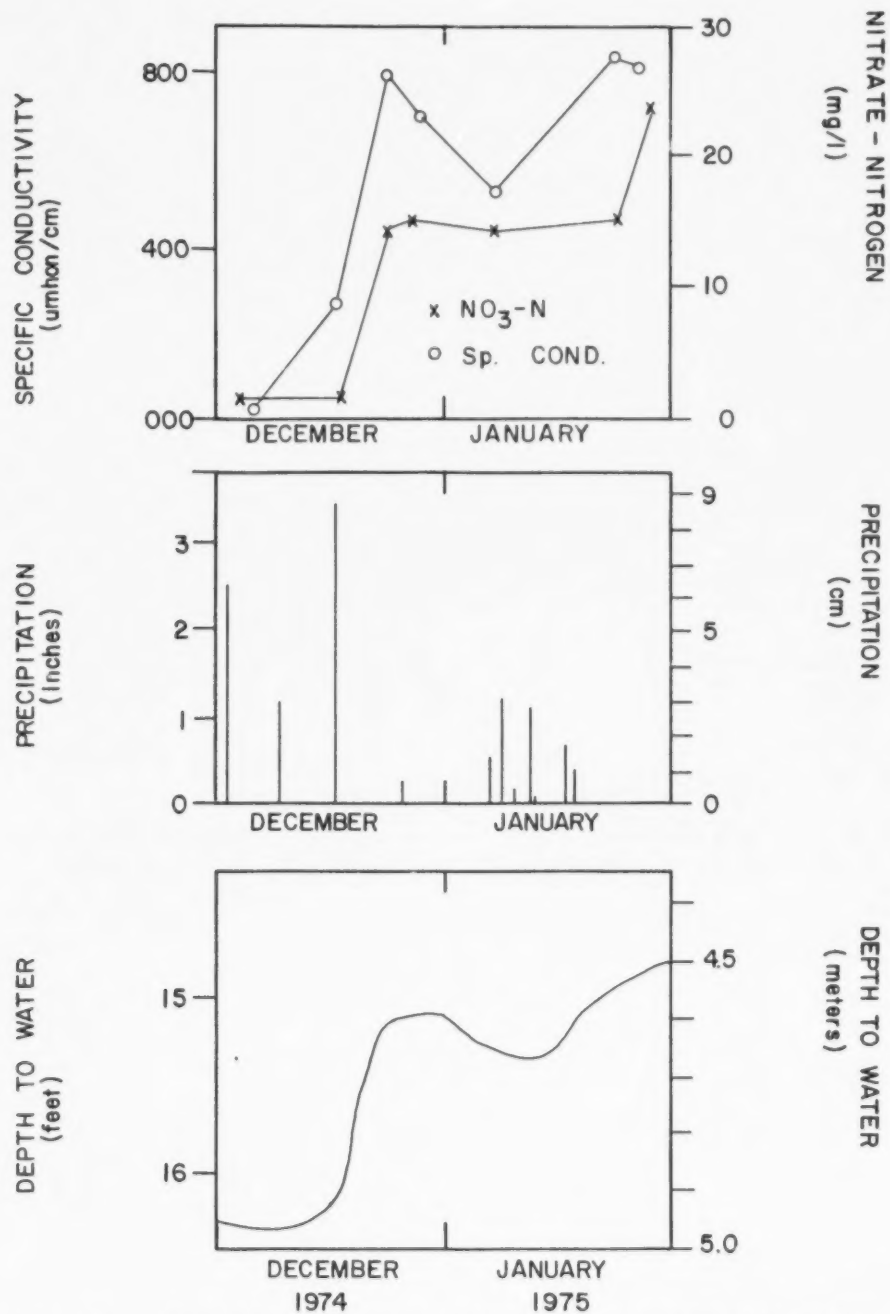


Figure 34. Specific Conductivity and Nitrate-Nitrogen response for Well WM21C as Related to precipitation and fluctuation in groundwater table.

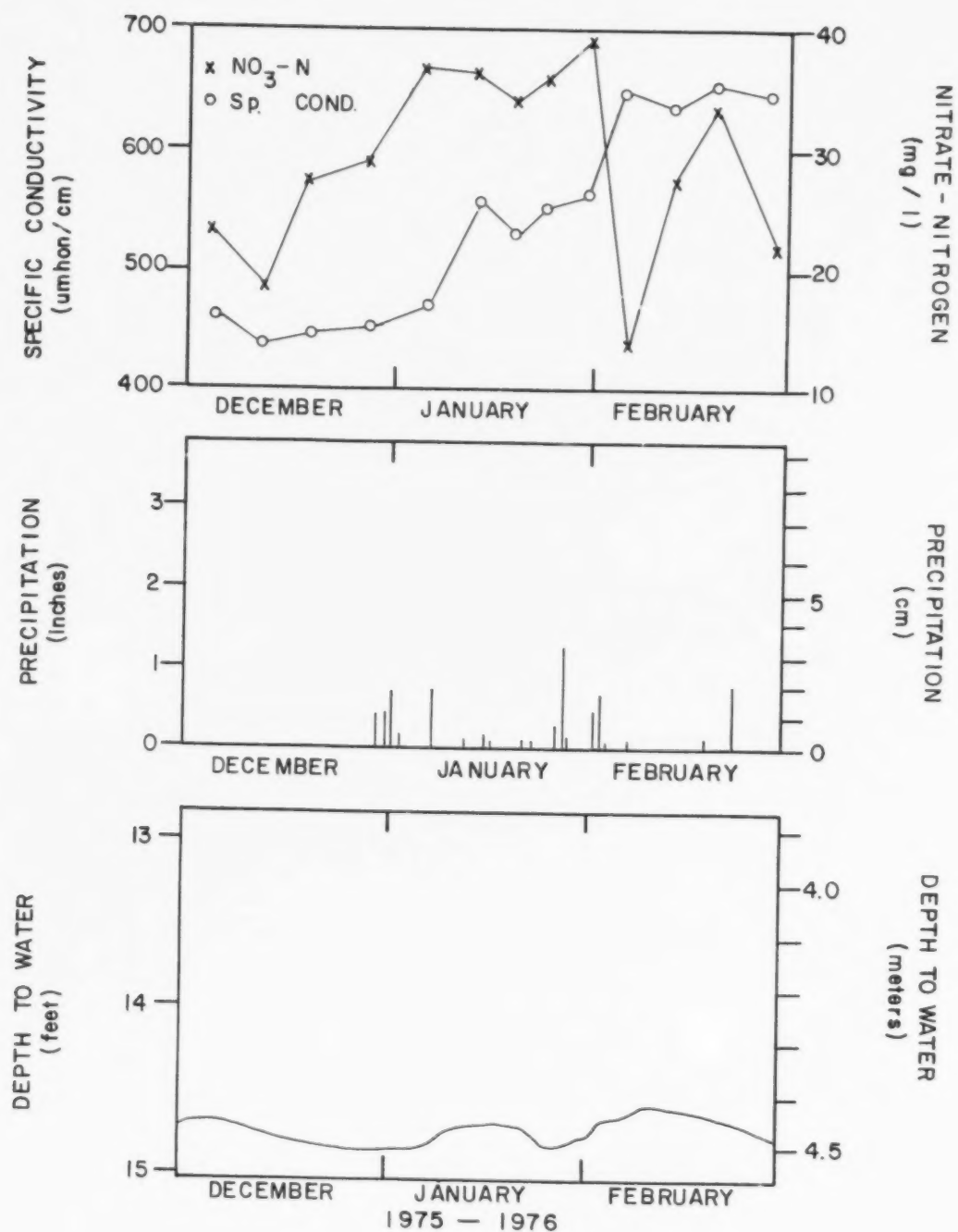


Figure 35. Specific Conductivity and Nitrate-Nitrogen response for Well WM12B as related to precipitation and fluctuation in groundwater table.

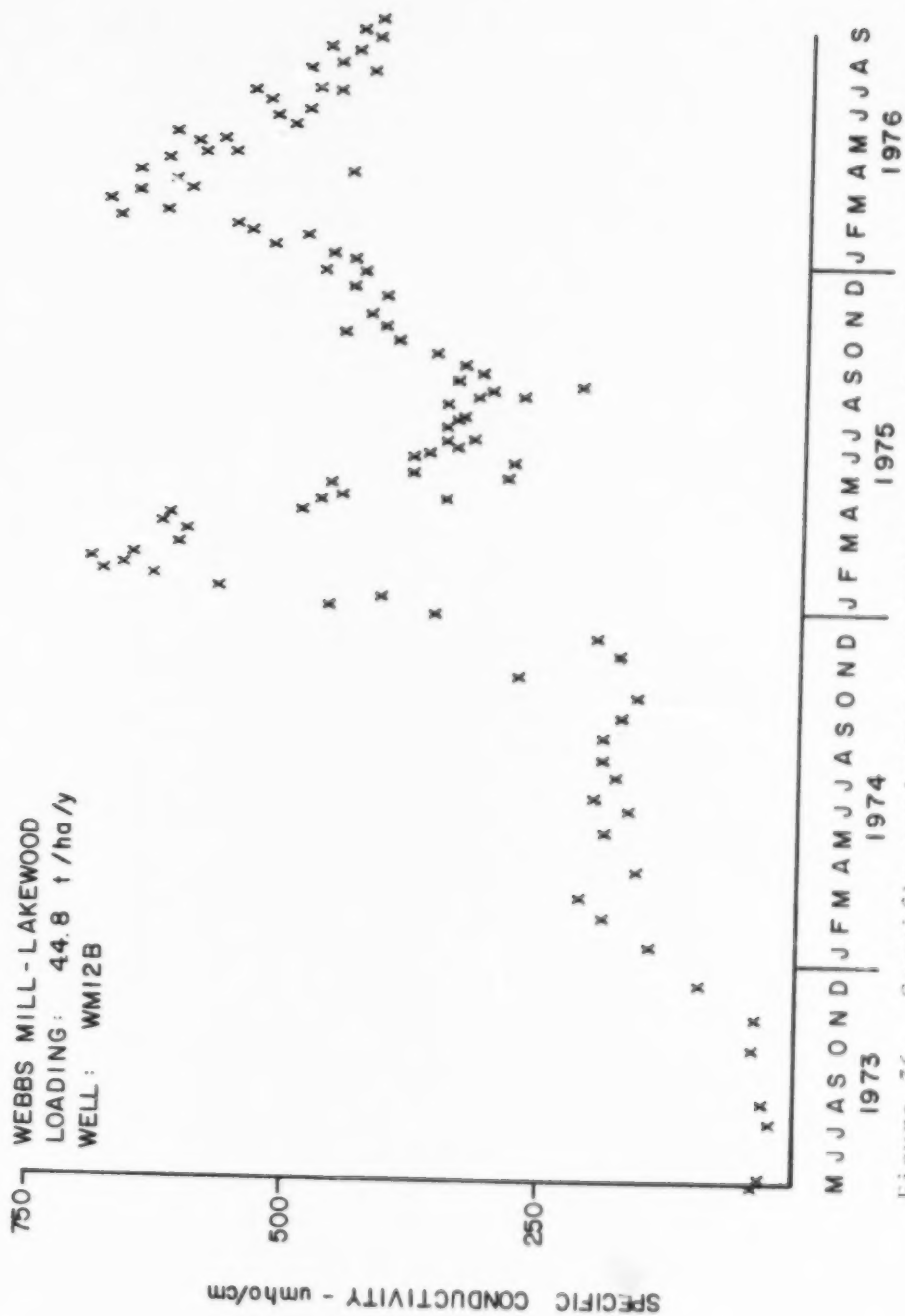


Figure 36. Specific conductance response over time for groundwater monitoring well on Webb's Mill Lakewood 44.8 t/ha/y plot.

TABLE 26. AVERAGE INCREASE OF CONTAMINANT BREAKTHROUGH OVER BACKGROUND LEVELS OF SELECTED PARAMETERS

Plot Location	Plot Loading (t/ha/y)	T.O.C.	NO ₃ -N	Parameters			
				NO ₂ -N (Increase in mg/l)	NH ₃ -N	SO ₄	Cl
Colliers Mills Downer	22.4	0.6	2.0	0.0	0.0	6	5
	44.8	0.4	5.4*	0.0	0.0	28	8
	44.8#	1.1	14.7**	0.0002	0.05	23	15
	89.6	1.9	23.8**	0.0004	0.06	72	25
Webbs Mill Lakewood	22.4	0.1	3.7	0.0	0.02	15	3
	44.8	2.2	22.7**	0.0	0.08	75	17
	44.8#	.4	4.3	0.0	0.0	7	5
	89.6	5.8	54.0**	0.003	0.13	123*	44
Webbs Mill Woodmansie	22.4 (B)	2.5	11.6**	0.0002	0.04	52	11
	(C)	0.6	4.1	0.0002	0.	11	1
	44.8	0.6	5.5*	0.0	0.06	9	3
	44.8#	6.7	26.7**	0.0016	2.20*	63	38
	89.6	.9	13.2**	0.0003	0.03	31	8

Natural Vegetation Plot.
 (B), (C) Well Designation for Wells 'B' and 'C' on 22.4 t/ha/y Woodmansie Plot.
 ** Increase is at or above applicable Drinking Water Standards.
 * Increase is within 50% of applicable Drinking Water Standards.

TABLE 26. AVERAGE INCREASE OF CONTAMINANT BREAKTHROUGH OVER BACKGROUND LEVELS OF
SELECTED PARAMETERS (Cont'd)

Plot Location	Plot Loading (t/ha/y)	Parameters Ni (Increase in mg/l)				
		Cu	Zn	Cd	Hg	
Colliers Mills Downer	22.4	0.0	0.02	0.0	0.0006	0.0
	44.8	0.002	0.03	0.0007	0.0004	0.0
	44.8#	0.001	0.07	0.0017	0.0013	0.0
	89.6	0.002	0.05	0.0028	0.0012	0.0
Webbs Mill Lakewood	22.4	0.001	0.01	0.0002	0.0003	0.0
	44.8	0.001	0.06	0.0005	0.0005	0.0
	44.8#	0.0	0.03	0.0	0.004	0.0
	89.6	0.160	0.17	0.0036	0.0007	0.0
Webbs Mill Woodmansie	22.4 (B)	0.0	0.04	0.0008	0.0005	0.0
	(C)	0.0	0.02	0.0003	0.0004	0.0
	44.8	0.0	0.01	0.0007	0.0002	0.0
	44.8#	0.004	0.25	0.0032	0.0031	0.0
	89.6	0.0	0.02	0.0018	0.0007	0.0

Natural Vegetation.

(B), (C) Well Designation for Well 'B' and 'C' on 22.4 t/ha/y Woodmansie plot.

** Increase is at or above applicable Drinking Water Standards.

* Increase is within 50% of applicable Drinking Water Standards.

TABLE 26. AVERAGE INCREASE OF CONTAMINANT BREAKTHROUGH OVER BACKGROUND LEVELS OF SELECTED PARAMETERS (Cont'd)

Plot Location	Plot Loading (t/ha/y)	T.D.S.	Parameters (Increase in mg/l)			
			Hardness	Ca	Mg	K
Colliers Mills Downer	22.4	19	10	1.4	1.5	2.0
	44.8	54	37	10.0	1.9	4.3
	44.8#	87	48	16.0	1.9	7.1
	89.6	330*	132*	37.0	9.0	10.5
Webbs Mill Lakewood	22.4	45	18	2.2	2.5	0.9
	44.8	145	125*	33.0	11.8	7.4
	44.8#	14	18	5.9	.7	2.0
	89.6	653*	289**	87.5	17.5	19.4
Webbs Mill Woodmansie	22.4 (B)	72	70	12.0	9.6	5.7
	(C)	-	16	1.7	2.8	0.8
	44.8	30	22	3.8	3.0	1.7
	44.8#	250*	122*	41.0	4.5	12.0
	89.6	92	60	18.0	3.5	4.4

Natural Vegetation.
 (B), (C) Well Designation for Well 'B' and 'C' on 22.4 t/ha/y Woodmansie plot.
 ** Increase is at or above applicable Drinking Water Standards.
 * Increase is within 50% of applicable Drinking Water Standards.

TABLE 26. AVERAGE INCREASE OF CONTAMINANT BREAKTHROUGH OVER BACKGROUND LEVELS OF SELECTED PARAMETERS (Cont'd)

Plot Location	Plot Loading (t/ha/y)	Parameters (Increase in mg/l)				
		B	Fe	Mn	Al	Pb
Colliers Mills Downer	22.4	0.0	0.0	0.12	0.0	0.0
	44.8	0.0	0.0	0.25	1.3	0.0
	44.8#	0.0	0.0	0.44	3.4	0.0
	89.6	0.01	0.0	0.66	4.6	0.0
Webbs Mill Lakewood	22.4	0.0	0.0	0.03	0.6	0.0
	44.8	0.0	0.01	0.05	4.0	0.0
	44.8#	0.0	0.0	0.03	1.4	0.0
	89.6	0.02	0.02	0.13	8.0	0.0
Webbs Mill Woodmansie	22.4 (B) (C)	0.0	0.0	0.08	0.6	0.0
	44.8	0.0	0.0	0.05	2.0	0.0
	44.8#	0.0	0.0	0.05	0.1	0.0
	89.6	0.01 0.0	0.08 0.01	0.33 0.10	11.5 1.6	0.0 0.0

Natural Vegetation.
 (B), (C) Well Designation for Well 'B' and 'C' on 22.4 t/ha/y Woodmansie plot.
 ** Increase is at or above applicable Drinking Water Standards.
 * Increase is within 50% of applicable Drinking Water Standards.

plot (Table 26). The 'B' well on this plot was installed in a pocket of sand not characteristic of the Woodmansie soil series (no clay or silt lenses at lower depth), while Well 'C' was drilled in a soil typical of the Woodmansie series. Therefore, the data corresponding to Well 'C' for the 22.4 t/ha/y Woodmansie plot is more representative than the Woodmansie series, while data reported for the 'B' well would be characteristic of a very sandy soil. Two additional distinctions pertaining to groundwater quality are also identified in Table 26: (1) when the contaminant concentration is at or above the drinking water standard and (2) when the contaminant concentration is within 50% of the maximum limit.

Heavy metals did not appear to have a significant impact on groundwater quality. The average concentrations of all the heavy metals were generally equivalent to or slightly above the natural background levels. When a marginal increase in metal concentration was observed, particularly for zinc and copper (refer to the Lakewood 89.6 t/ha/y and Woodmansie natural vegetation plots), it was well below the maximum contaminant limits for potable water.

The comparison of contaminant breakthrough concentrations beneath individual plots to potable drinking water standards was performed utilizing the average contaminant concentration. The peak concentrations were not considered in this comparison. It appeared the average contaminant concentration was more representative of the long-term response of the groundwater system to contaminant breakthrough, while the peak concentration was only the maximum level observed at only one single point in time. Therefore, the average contaminant concentration was selected for the analysis.

No specific organic analyses were performed on the groundwater samples. Only the coarse measurement of total organic carbon was obtained for each sample. Total organic carbon was found to have increased above background level beneath many of the plots. The breakthrough of organic carbon as measured by TOC tends to indicate that trace organics may have entered the groundwater. This is particularly true for those plots receiving the higher wastewater solids loadings, Lakewood 89.6 t/ha/y plot and where soil conditions are extremely acid as was the case with all the natural vegetation plots.

Nitrate-nitrogen was the limiting groundwater chemical constituent with respect to drinking water standards. Additionally, average breakthrough concentrations of total dissolved solids, chlorides, sulfates, and total hardness (calcium and magnesium) were approaching, and in many cases exceeded,

the recommended limit beneath the following plots: plots receiving 89.6 t/ha/y of wastewater solids, natural vegetation plots (Downer and Woodmansie) and where soil characteristics were similar to that of a coarse sand. For the Downer and Woodmansie soils, it would appear a surface wastewater solids loading rate in excess of 22.4 t/ha/y and possibly as much as 44.8 t/ha/y could be applied without violating the potable water standards downgradient from the application site. In the case of the more sandy Lakewood soil, the maximum allowable loading rate appears to be 22.4 t/ha/y. However, where the soil is characteristic of a coarse sand, containing little or no silt or clay lenses, even the 22.4 t/ha/y application rate could violate potable drinking water standards. It should be noted that the application rates suggested are based on contaminant levels directly beneath the application plot. They do not reflect the possible dilution downgradient of the application plot from the natural diffusion and dispersion of the contaminant in the groundwater.

Nitrate-Nitrogen Contamination

The land application of wastewater solids has resulted in groundwater contamination above baseline levels beneath almost all plots. However, the extent of contamination and possible groundwater pollution was found to be significantly dependent on wastewater solids, loading rates, soil character, and soil pH. The groundwater chemical constituent which caused the greatest concern was nitrate-nitrogen ($\text{NO}_3\text{-N}$). The nitrate-nitrogen breakthrough concentrations beneath most application plots receiving the larger wastewater solids loadings were in violation of the present Federal drinking water standard of 10.0 mg/l $\text{NO}_3\text{-N}$. Nitrate-nitrogen levels ranged as high as 102 mg/l at the Webbs Mill-Lakewood 89.6 t/ha/y plot.

The 'D' and 'E' wells were installed downgradient of the application site according to the original potentiometric maps determined at the beginning of the study in 1973. However, it was found that the direction of the flow shifts under changing conditions of flow gradients, recharge and discharge. Therefore, the positions of several downgradient profile samplers were relocated to identify the zones of contamination within the groundwater system. Thus, the concentration of nitrate-nitrogen reported for these wells at the water table level were suspect due to changing flow direction. The result was a value considerably lower than the representative concentration which lies directly along the line of flow. Skim-samples withdrawn from downgradient wells were also somewhat diluted because those wells were not subject to vertical nitrate breakthrough, but are subject to groundwater recharge; hence, dilution. Additionally, the peak and average $\text{NO}_3\text{-N}$ concentrations reported do not identify significant dynamic trends

within the groundwater system; i.e., increase or decrease in $\text{NO}_3\text{-N}$ levels as contaminant travels through the system.

Average nitrate-nitrogen concentrations beneath and downgradient of the Colliers Mills-Downer and Webbs Mill-Lakewood 22.4 t/ha/y application plots were less than 4 mg/l. However, directly beneath the Webbs Mill - Woodmansie 22.4 t/ha/y application plot, $\text{NO}_3\text{-N}$ concentrations were considerably greater. The peak $\text{NO}_3\text{-N}$ concentration was 38 mg/l; and the average $\text{NO}_3\text{-N}$ concentration was in excess of 10.0 mg/l. Downgradient from the Woodmansie 22.4 t/ha/y plot levels of nitrate-nitrogen contamination were minimal compared to concentrations beneath the application plot. This skewed effect among plots was the result of an atypical pocket of sand located in the center of the Webbs Mill - Woodmansie 22.4 t/ha/y plot.

Nitrate-nitrogen concentrations beneath the cropped plots receiving a wastewater solids loading of 44.8 t/ha/y were considerably greater at the Webbs Mill-Lakewood plot than at the other two (2) locations. The peak and average nitrate-nitrogen concentrations at the water table level were 79 and 31 mg/l, respectively. Nitrate breakthrough levels at the Colliers Mill-Downer and Webbs Mill-Woodmansie 44.8 t/ha/y plots were generally less than 10 mg/l $\text{NO}_3\text{-N}$. Although breakthrough concentrations may have been less than 10.0 mg/l $\text{NO}_3\text{-N}$ beneath these plots, profile samples tended to indicate that significant nitrate-nitrogen contamination above the 10.0 mg/l level had occurred downgradient of these application plots.

Considerable $\text{NO}_3\text{-N}$ contamination occurred at the natural vegetation plots. $\text{NO}_3\text{-N}$ breakthrough ranged as high as 89 mg/l beneath the Webbs Mill-Woodmansie plot. Downgradient contamination was also substantial at the Colliers Mill-Downer and Webbs Mill-Lakewood natural vegetation plots. For example, $\text{NO}_3\text{-N}$ levels were as high as 32 and 30 mg/l at distances of 30.5m and 61m downgradient of the Colliers Mills-Downer plot, respectively.

Nitrate-nitrogen contamination was the greatest at the application plots receiving a wastewater solids loading of 89.6 t/ha/y, particularly at the Webbs Mill-Lakewood site. Nitrate-nitrogen levels beneath the Webbs Mill-Lakewood 89.6 t/ha/y plot ranged as high as 102 mg/l. Downgradient $\text{NO}_3\text{-N}$ from the values reported directly beneath the plot. Peak $\text{NO}_3\text{-N}$ concentrations 30.5m and 61m downgradient from the application plot were 96 and 74 mg/l, respectively.